Climate Change Impacts on Tropical Forests in Central America

AN ECOSYSTEM SERVICE PERSPECTIVE

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EDITED BY ALINE CHIABAI

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The loss of biodiversity is a major environmental problem in nearly every terrestrial ecosystem on Earth. This loss is accelerating driven by climate change, as well as by other causes including agricultural exploitation, fragmentation and degradation triggered by land use changes. The crucial issue under debate is the impact on the welfare of current and future population, and the role of humans in the exploitation of natural resources. This is of particular importance in Central America, which is among the richest and most threatened biodiversity regions on the Earth, and where the loss of ecosystems strongly affects its socio-economic vulnerability.

This book addresses the impacts of climate and land-use change on tropical forest ecosystems in this important region, and assesses the expected economic costs if no policy action is taken, under different future scenarios and for different geographical scales.

This innovative collection utilizes both theoretical approaches and empirical results to provide a conceptual framework for an integrated analysis of climate and land-use change impacts on forest ecosystems and related economic effects, offering insight into the complex relationship between ecosystems and benefits to humans. This important contribution to forest ecosystems and climate change provides invaluable reading for students and scholars in the fields of environmental and ecological economics, environmental science and forestry, natural resource management, agriculture and climate change.

Aline Chiabai is a senior researcher at the Basque Centre for Climate Change (BC3), where she coordinates the research area on health and climate change.

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Climate Change Impacts on Tropical Forests in Central America
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Edited by Aline Chiabai

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Edward B. Barbier, University of Wyoming, USA

"Tropical forests are among the most valuable ecosystems on earth in terms of their biodiversity and their contribution to sustainable human well-being. This book is a timely and valuable addition to the growing literature on tropical forest ecosystem services."

Robert Costanza, The Australian National University, Australia

"This is a worth reading volume for researchers and policy makers which presents one of the few ecological and economic comprehensive studies of how climate change is affecting ecosystem services. The methodological approach and analysis, combining different types of spatially explicit data within various models and scenarios, should serve as a model for other critical regions."

Maria Jose Sanz Sanchez, Agriculture Organization of the United Nations (FAO), Italy

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Introduction

Aline Chiabai

The loss of biodiversity is a major environmental problem in nearly every terrestrial ecosystem on Earth. This loss is accelerating, driven by climate change, as well as by other causes including agricultural exploitation, fragmentation and degradation triggered by land-use changes. In particular, the impact of climate change on forest ecosystems and biodiversity is well documented in the literature and supported by the recent Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report (IPCC 2014), which states that irreversible modifications in the functioning and structure of forests are predicted to occur, especially in the Amazon and the Arctic. The expected changes in temperature and precipitation patterns will cause an increase in the intensity and frequency of extreme weather events, an expansion of subtropical desert regions, changes in agricultural yields and in the geographical distribution of disease vectors. All these factors will have profound impacts on the geographical distributions of natural communities, triggering modifications in plant and animal species, and increasing the risk of extinction.

These impacts are of particular importance in Central America where, on the one hand, a major share of the world's biodiversity capital is concentrated and, on the other hand, ecological biodiversity loss strongly affects socio-economic vulnerability. Central America also represents an important biological corridor serving as a bridge between the North and South American ecosystems. Maintaining connectivity is therefore crucial for resident species of conservation concern that annually migrate in response to annual changes in the wet and dry seasons. The narrow, north–south orientation of the Central American landmass makes it particularly vulnerable to climate change. At its narrowest point, the isthmus is less than 130 km wide. Deforestation and loss of habitat in the isthmus can have consequences for biodiversity that are felt beyond the region. The current rate of the anthropogenic change, including the burning of greenhouse gases and agricultural expansion, is creating a doubly perilous situation threatening regional and global biodiversity. Protecting biodiversity and ecosystem services in the face of climate change must therefore be a key priority of this century.

The potential consequences of biodiversity loss on the society have stimulated considerable debate in recent years, as this is affecting ecosystem functioning and the goods and services provided to humans. As biodiversity decreases, we are losing

species and varieties of species, as well as goods and services to humans. The crucial issue under debate is the impact on the welfare of current and future population, and the role of humans in the exploitation of natural resources.

Expected impacts of climate change on ecosystem services will therefore affect human health both directly and indirectly. Services such as water supply, recreational activities, disease regulation and bioprospecting have a straightforward impact on human health, while others, such as soil retention and climate regulation, affect health through indirect pathways by providing a healthy environment and mitigation opportunities. Forest fires, as well as deforestation and fragmentation due to land-use change, will act as additional stressors to climate change by increasing the risk of degradation of tropical forests and causing a decline in many services. Degradation will result in deterioration of water quality, disruption of water supply, soil erosion, with increased risks of droughts or flooding, and infectious disease outbreaks. Water supply is among the forest services that will be highly affected by climate change, and freshwater ecosystem will see the highest amount of species at risk of extinction (Settele *et al.* 2014; MEA 2005). Central and South America show a high reliance on water supply which is critical for economic sectors such as agriculture and hydropower. Expected reduction of water flows and water quality would therefore have a high impact on the national economies as well.

All these impacts will affect human welfare and can be translated in a cost for the society through a variety of techniques known as market and non-market valuation, depending on the type of service addressed. Some services, such as recreational activities or aesthetic view are more difficult to value as they are not traded in the market and do not have therefore a price of reference, in comparison with other services such as hydropower and timber. All in all, the importance of economic valuation is incontestable for making appropriate decisions in the face of climate change and to set up suitable mitigation and adaptation strategies. Estimates of the cost of inaction, which is the expected impact without policy action, are also critical for policy-makers in the choice of future paths of development.

In this context a number of studies exist in the literature addressing the bio-physical impacts of climate change on tropical forests in Central America (Diaz and Morehouse 2003; Duncan 2011) or in other regions (e.g. Reich and Oleksyn 2008; Adams *et al.* 2009), while others are focusing more specifically on the expected economic costs for the society (Lawford, Alaback and Fuentes 1995; World Bank 2009; Gan, Grado and Munn 2010; Ding 2012). However, the coverage of available studies in the ecological and economic areas is still incomplete, and further research is required on both sides, as well as on their integration into a common framework suitable to link climatic and bio-physical processes with economic valuation and social welfare. Indeed, the role of climate change in this context is far from being understood, as well as the relationship between natural ecosystems, the provision of ecosystems services and their benefits to humans (TEEB 2010). Current research developments are also calling for studies that investigate on the economic values of ecosystem services under climate change.

This book is a response to this research need and an attempt to provide more insights on this complex relationship through a multidisciplinary collaborative effort among environmental, bio-physical, and socio-economic scientists. It is the result of a collaboration between the Basque Centre for Climate Change (BC3) (Spain) and the Centro Agronómico Tropical de Investigación y Enseñanza (CATIE) (Costa Rica), within the two-year project "Climate Change and Biodiversity Loss: The Effects on Ecosystem Services" (CLIMBE), financed by FBBVA and carried out in the period 2010–2011.

The project assessed the impacts of climate and land-use change on tropical forest ecosystems services in Central America, and projected the economic costs for the society under different future scenarios and for different geographical scales. The proposed methodological approach combined climatic, bio-physical, and economic data within a spatial framework using various models and scenarios. Climate change and land-use change, considered as the two most important stressors for biodiversity loss (Sala *et al.* 2000), were analyzed within different temporal and spatial scales separately. Central America was chosen as the study area since it is one of the richest and highest threatened biodiversity regions on the planet, and because the loss of ecosystems and related services strongly impacts socio-economic vulnerability. Estimates of the cost of inaction are therefore of considerable interest to assist the design of effective conservation policies and decide future paths of economic development.

The book is structured into three parts. In particular, the first two parts present the theoretical framework and methodological approaches developed to assess the impacts within CLIMBE project. The third part explores the role of economic valuation and ecosystem-based adaptation in a policy context. The content of each chapter, approaches used and synthesized results achieved are discussed in the following sections.

Part I: Central American tropical forests, ecosystem services and human wellbeing

The first part of the book offers to the reader a general overview of the existing tropical ecoregions in Central America, the types of services they deliver and the main threats and degradations they are facing nowadays. It also introduces basic concepts linked with ecosystem services, economic values, and human wellbeing.

More specifically, Chapter 1 provides a discussion of the current status of tropical forests in Central America, and their characterization in terms of bio-physical environment, forest types, land cover, climatic regimes, biodiversity levels, and major ecological processes. Existing threats leading to degradation and conservation purposes are also discussed. The analysis is based on the construction of a spatial database at regional scale and is supported by the existing literature in this context. The study makes use of the concept of "ecoregion," identified by vast geographical areas of land and water which may contain different ecosystems, and defined as "the areas within which there is spatial coincidence in characteristics of geographical phenomena associated with differences in the quality, health and

integrity of ecosystems" (Omernik 2004). This concept is useful as it helps in categorizing and comparing representative habitats.

Tropical forest in Central America is recognized as the fifth world's hotspot, for being one of the richest areas of the Earth in terms of species diversity (Conservation International 2011). The chapter shows that higher species richness of macro-fauna is recorded in the southern wet forests, while the driest areas show higher levels of endemic species. Each ecoregion provides a set of "marketable and non-marketable ecosystem services." For example the montane forests are characterized by the importance of hydrological services, while the humid forests produce high levels of organic matter. Land-use change is identified among the main causes for habitat losses in Central America, and climate change would be an additional driver magnifying expected impacts. In fact, in the tropical areas, Central America is considered the region with the highest expected change in climate (Giorgi 2006). Imbach *et al.* (2012) found that in the period 2070–2100 the leaf area index could be reduced by 77–89 per cent producing a big reduction in tree cover and a conversion of tropical rainforests to dry forests. Kannan and James (2009) predicted significant changes in precipitation patterns with strong impacts on the hydrological services. Among ecosystems, the freshwater system is expected to see the highest threat of extinction caused by global warming (MEA 2005), due to the increase in water temperatures, modification of water flows and loss of habitat. All these changes in the habitat conditions will affect the provision of ecosystem services as well as resilience to climate change.

Protected areas have been established for conservation purposes, but they are usually quite small in extension, isolated, and affected by the development of economic activities, mainly agriculture and pasture (DeClerk *et al.* 2010). They cover different portions of the territory, from a minimum extension of 1 per cent of the territory in Chiapas to a maximum of 66 per cent in the Talamancan montane forests. Given the many threats in the region, the chapter highlights that conservation efforts should be intensified and consolidated through the establishment of protected areas and biological corridors to promote biological diversity and connectivity. Protected areas alone will be, however, insufficient to preserve ecosystem services and to cope with the extending degradation of natural areas. Restoration of natural habitats is also critical as an adaptation measure in the region, as it can contribute to the economic development of rural population, as well as to improve resilience to climate change.

Chapter 2 discusses the relevance of tropical forest ecosystem services for human health and wellbeing, their utility in national economies, how their value can be translated into economic units and to what extent economic valuation can be a valid tool for policy-makers. The chapter begins by providing a background on the main threats and drivers of changes for Central American tropical forests and how these changes will affect the provision of ecosystem services and human wellbeing. The link with human health is also discussed considering both direct and indirect pathways of propagation. Climate change impacts are analyzed within the IPCC Fifth Assessment Report (IPCC 2014) and interactions with other causes are contextualized. The chapter continues by describing the types of

ecosystem services relevant for tropical forests and specifically for Central America, together with their links with economic sectors and national economies. Problems associated with existing categorizations of ecosystem services are discussed in the context of economic valuation. Monetary values are critical for informing decision-making processes in the allocation of financial resources among different objectives, and to value the appropriateness of mitigation and adaptation plans. There are, however, a number of key issues which strongly affect the results and convenience of the economic assessment. These include, among others, the presence of intangible and intrinsic benefits in the ecosystem which are hardly quantifiable, the dependence of economic values on local socio-economic contexts, the non-linear relationship between forest loss and loss of services flows, as well as issues related to the commodification of nature and ecological thresholds. All these aspects call for more inclusive approaches able to incorporate and combine different techniques and the manifold dimensions characterizing the provision of ecosystem services, such as social and collective values (TEEB 2010). Some attempts exist in the literature such as the development of human wellbeing measures (Villamagna and Giesecke 2014), or the combination of deliberative processes with economic valuation within participatory models trying to capture the multiple dimensions of the service value, though these approaches suffer from other limitations and represent a limited response to the acknowledged inadequacies of economic values. Other examples include the idea of focusing on beneficiary groups and mapping the spatial dynamics of ecosystem service flows (Villa *et al.* 2014), but these approaches lack the inclusion of appropriate monetary assessment. Future research should explore the conditions for a full integration of ecologic and economic approaches in order to address the current limitations of monetary valuation and characterize the formation of economic values within a wider context able to link these values with the quality of the ecosystem. In this respect, this book is an attempt to tackle current issues associated with methodologies and estimation processes, in view of integrating ecological and economic science, and rethinking main theoretical approaches.

Part II: Climate, water and land-use changes in Central American tropical forests

Four chapters are included in the second part of the book, reporting on the methodologies and results vis-à-vis the assessment of the impacts of climate and land-use changes, which are two major stressors for biodiversity loss in Central America operating at different temporal and spatial scales. A bio-physical model is used to assess the impacts of climate change on ecosystem services flows at regional scale for the whole Central America by the end of the century. On the other side, a land-use change model explores how changes in land uses can affect the provision of ecosystem services in a specific biodiversity hotspot and within a shorter temporal scale. In both cases, bio-physical outputs are integrated into economic models in order to translate the ecological impacts into monetary

impacts. The economic tools available from the literature are those discussed in Chapter 2.

Chapter 3 focuses on the hydrological services and discusses the use of the biophysical model MAPSS (Mapped Atmosphere Plant Soil System) to estimate the expected changes in water availability in the principal watersheds at regional scale, under the intermediate emission scenario RCP 4.5 (Representative Concentration Pathways) for the year 2050 (2041–2060) and the year 2070 (2061–2080). MAPSS is a soil–vegetation–atmosphere transfer (SVAT) model, usually used to simulate ecosystem functioning in the interactions between climate, vegetation and soils. The model accounts for complex non-linear interactions between vegetation dynamics and water balance, assuming potential vegetation cover. This highlights the importance of accounting for changes in vegetation when assessing impacts on the water balance under climate change, as opposed to traditional hydrological modeling approaches where the vegetation is usually forced to be constant. The model is built on potential vegetation, meaning that land-use changes are not reflected in the analysis, which might appear as a limitation at first glance, though the authors (Imbach *et al.* 2010) show that in the long run there is no clear effect of land-use changes on runoff at the regional scale of Central America.

Future scenarios for Central America indicate a general increase in mean temperatures for scenario RCP 4.5 (ranging between 1.4°C and 2.2°C for 2050, and 1.8°C and 2.6°C for 2070), while predictions for precipitation exhibit a decreasing pattern in the northern regions (from Guatemala to Nicaragua) and a rising trend in the South (Costa Rica and Panama), although with larger uncertainties for the latter. Regarding the water balance, the model shows a general drying trend in the whole region, with runoff expected to decrease in 81 per cent of the countries by 2050 (and 89 per cent by 2070), especially in the north (Honduras, El Salvador and Guatemala). Potential vegetation will show a shift from humid to dry species, and runoff is likely to be reduced even under scenarios showing a precipitation increase due to evapotranspiration intensified by the rise in temperature (Imbach *et al.* 2012).

In summary, these impacts will translate in an increase in the population under pressure for water scarcity. In year 2050, the number of people with limited access to water resources will increase from 6 per cent in the current situation to 26 per cent, while the population under scarcity condition will reach 15 per cent. In 2070, the situation will further aggravate, with 21 per cent of the population expected to be under scarcity conditions and 18 per cent under absolute scarcity. Finally, strong impacts are expected in economic sectors which are highly dependent on water resources such as hydroelectricity and agriculture.

Hydrological services are given a special focus in the book due their critical importance in the countries of Central America for safe drinking water, hydropower and agriculture, and given the expected changes induced by climate which could strongly affect their provision (Locatelli *et al.* 2010). Water demand is indeed expected to increase under future scenarios (given the pressure of industrial development and population growth) vis-à-vis a decrease in water supply. Knowing the impacts of climate change in this context is of crucial importance for decision-

making about water management, in order to plan adequate adaptation strategies to address the needs of communities, farmers, hydropower plants and industries in general.

Given the expected reduction in water supply under future scenarios, Chapter 4 presents a methodological approach to assess the climate change impacts on the hydropower sector for Costa Rica specifically, where hydroelectricity plays a crucial role in the national economy, contributing to a large extent to the total energy produced in the country. The method combines the bio-physical model MAPSS with an economic model based on production function and spatial referenced data, to project the expected changes in hydro-energy generation by 2100, as a result of reductions in annual runoff for the IPCC scenarios A2, A1B, and B1.

The production function proposed in this chapter relates the revenues produced by each plant with the expected changes in runoff and a series of technical variables characterizing the plant. A change in the runoff of a watershed has, indeed, a direct impact on the production of electricity and consequently on the expected revenues.

The theoretical model is contrasted with previous studies in the literature and consequently applied to a sample of 35 hydropower plants, producing more than 70 per cent of the total hydroelectricity generated in Costa Rica in 2009 (CEPAL 2009).

The economic impact, calculated on the expected revenues for the sample under analysis, ranges from 5 per cent loss in the environmentally driven scenario (B1) to 9 per cent and 12 per cent in the economically driven scenarios (A1B and A2 respectively). Strong variations among plants might be envisaged, with some of the plants displaying losses above 30 per cent in all scenarios.

In spite of the limitations, which are thoughtfully discussed in the chapter, the model shows the importance of factors such as runoff and installed capacity in influencing the production of hydroelectricity and its revenues, while other technical features are less critical.

A strong scientific point in the study is the liaison between the bio-physical and the economic modelling, and the prospect of using the model for projections under climate change and scaling-up to wider geographical scales. This liaison is done by mapping runoff data by watershed for each hydropower plant and incorporating them into a spatially referenced database for the production function, combining economic values with spatial information.

The main policy contribution of this study lies in the potential of the model to identify the hydropower plants having the highest vulnerability to climate change. Given the significant reduction expected for water supply in Costa Rica as a result of climate change, and the importance of the hydropower sector at national level, the definition of adaptation strategies in this context is crucial and should be prioritized. Besides the opportunity to consider other alternatives for energy supply, possible technological solutions include the need to increase the operating efficiency of the existing plants, to promote the design of new infrastructures less vulnerable to climate, and to enhance the management of the reservoirs to optimize water flows and compensate periods of lower runoff.

Chapter 5 focuses on services which are not traded in regular market places and therefore often neglected in policy decisions. The services under analysis include specifically recreation and water. The methodological approach combines economic and vegetation models to assess the economic impacts of climate change on these services in the Central American tropical forests by 2100.

Recent trends in economic valuation of ecosystem services have shown the need to address the bio-physical foundation and the ecological processes behind the provision of services and their economic values (Polasky 2009). In this context, similarly to Chapter 4, the main contribution of this chapter lies in the integration between these two aspects through the construction of a spatially referenced database with bio-physical information estimated at the site level. More specifically, data estimated through a vegetation model based on Holdridge zones are integrated within an economic model centered on spatially-referenced meta-analysis, benefit transfer and up-scaling techniques.

Not being traded in the market, the services analyzed in this chapter are assessed using non-market values taken from the existing literature in the field (stated and revealed preferences), as well as values generated within payment for ecosystem services (PES) schemes.

The chapter concludes that future expected changes in forest size and vegetation types in Central America will strongly affect annual economic benefits which are likely to decrease, though the specific impacts differ among geographical region and forest type. Wet and rain forests are those suffering from larger economic impacts, principally due to the loss of recreational services which are of particular importance in these forests. Impacts are predicted to be substantial even in the most sustainable and environmentally-driven scenarios, though the largest impacts are expected in A1 and A1B scenarios. In this context, mitigation and adaptation options are crucial to protect the forest ecosystem in Central America. Results obtained in this chapter highlight the importance of implementing strategies for conservation with a specific target on wet and rain forests as they are the most threatened ecosystems and they provide services with the highest economic values.

The last chapter in Part II (Chapter 6) analyses the impacts that land-use changes can have on the provision of ecosystem services and on local livelihoods, and proposes a methodology to assist decision-makers in the identification of the most sustainable land-use management strategies, taking into account the interests of different local stakeholders. The proposed land-use strategies include some practices relying on ecosystem-based adaptation (the latter being extensively discussed in Chapter 8).

The analysis is performed at a local level for a biodiversity hotspot located in Costa Rica, the Volcánica Central Talamanca Biological Corridor (VCTBC). The VCTBC was created within a larger conservation strategy called Mesoamerican Biological Corridor, a regional initiative aiming at protecting biological diversity, as well as representative ecosystems in a way that both social development and economic sustainability could be achieved. Nowadays, the conservation of biodiversity in protected areas in Costa Rica is increasingly

becoming integrated in the agricultural matrix to provide secondary habitat and connectivity between reserves. Therefore, investigation on both economic and ecological aspects of land-use changes in the corridor is essential for the design of cost-effective management strategies to reduce ecological and human vulnerability.

Chapter 6 presents a methodological framework where bio-physical indicators estimated through a land-use change model are integrated within a social cost-benefit analysis in order to predict economic impacts in the period 2010–2030. The proposed temporal scale is shorter than in the analysis performed in the previous chapters, as specific and concrete land-use changes can be reasonably predicted only on a relatively short period of time.

The study considers a wide range of ecosystem services provided in three land-use scenarios having different degrees of environmental sustainability and economic development. Scenarios incorporating higher degrees of sustainability criteria contemplate conservation plans of natural forest and agricultural practices such as agroforestry and silvopastoral systems, which turn out to have a positive impact on the provision of ecosystem services, as well as increased landscape connectivity and improved biodiversity levels. The net present value in the strong sustainable development future scenario is estimated to be 17.5 per cent higher than in the status quo, while choosing the intensive economic future scenario would cause the deforestation of 4.5 per cent of natural forests, entailing a 6 per cent decrease in carbon stocks and consequently a 6 per cent reduction in total forest values. These losses are estimated to attain over US$400 million by 2030 in the VCTBC.

The chapter concludes that the social benefits generated by the strong sustainability scenario could be used to compensate the farmers for the initial economic costs of shifting from traditional forms of agriculture into more sustainable practices. In the long run, the new systems are able to increase the economic benefits of farmers due to increased land productivity and additional revenues from timber products for example. In this context, PES schemes have been extensively used in Costa Rica to encourage environmentally friendly practices and to promote forest conservation. A reallocation of resources towards sustainable land management calls, therefore, for innovative economic instruments able to incorporate conflicting stakeholders' interests. New business models, such as agro-tourism, habitat banking or pre-biodiversity certified products, could be seen as a possible response to the growing interest of establishing more sustainable development paths.

Part III: Economic assessment, adaptation options and policy implications

The third part of the book contains two chapters aiming at discussing more explicitly some policy-relevant issues, such as the use of ecosystem services valuation in a global and interdisciplinary context, and the application of ecosystem-based adaptation as a natural response to the expected impacts of climate change on tropical forests.

Chapter 7 reflects upon some policy-related aspects associated with the use of economic valuation as a decision-making support tool for conservation and adaptation purposes. The chapter discusses the factors that are expected to influence the economic values of ecosystem services in the medium-long term, as well as the main implications for adaptation in a context of large uncertainties as predicted in the current models. The impacts estimated in previous chapters are examined in a broader context of socio-economic changes at global scales and for Central America.

Economic valuation of ecosystem services is relevant for appropriate decision-making about the use of natural resources, but a number of issues may affect predictions for the future. These include the need to know the economic and social development paths that countries will follow in the long-term future, their socio-economic structure in terms of income and demographic projections, as well as implications in terms of urbanization, migration and energy demand. All these aspects might be crucial in a world of uncertainty, which characterizes not only the climatic and bio-physical impacts, but also the socio-economic contexts. Uncertainties in the latter could be even higher than the previous, and values of ecosystem services would be consequently affected, in terms of higher variability of expected impacts. Given these uncertainties, evidence from the literature recommends the implementation of measures known as no-regret or low-regret options which will bring benefits even in the absence of climate change and can considerably reduce the impacts at a low cost. In addition, these measures are able to provide substantial co-benefits for vulnerable groups and in different sectors outside the mere adaptation context.

Finally, Chapter 8 discusses the use of ecosystem-based adaptation (EbA) as a way of strengthening resilience and reducing vulnerability to climate change. The chapter presents three examples of existing best practices carried out in Central America to reduce the impacts on carbon stocks, water supply and recreation. It is worthwhile noting that most of the best practices have been recorded outside European countries, though the latter are increasingly engaged in projects based on ecosystem improvement.

In a nutshell, EbA refers to cost-effective measures aiming at protecting, maintaining, restoring and managing the ecosystems, to help people adapt to the changing climate while sustaining their livelihoods by improving the supply and quality of ecosystem services. These measures are characterized by a high degree of flexibility as they use the "buffer-capacity" of the natural environment, and they are more effective and less costly than engineering solutions based on grey infrastructures.

Inter alia, EbA interventions improve the capability of forests to sequester and store carbon, and if we consider that terrestrial ecosystems and oceans absorb about half of anthropogenic CO_2 emissions (Cannedell *et al.* 2007), the critical importance of these measures for mitigation can be easily put into perspective. Similarly, EbA helps in regulating the water cycle, improving water quality and soil retention, and limiting therefore the impacts of tropical storms and extreme events. It offers also opportunities to develop eco- and agro-tourism activities.

Besides the direct benefits associated with a reduction of climate change impacts on ecosystem services, these measures provide also many other co-benefits, such as increased values of biodiversity, health benefits due to improved contact with nature, promotion of more active lifestyles and mental health. All these benefits are immediately accessible to the rural communities.

Specific solutions for ecosystem-based adaptation include a large set of options such as sustainable agricultural and forest management practices (e.g. agroforestry and silvopastoral systems), re-forestation and restoration of degraded land, regulation of water flows and provision of clean water. Central America is, however, at a very early stage in the promotion of such practices. Barriers to implementation include, among others, the insufficiency of studies assessing the effects of land uses on the provision and quality of ecosystem services. In this respect, Chapter 6 is an attempt to provide a theoretical framework for such an appraisal, linking land uses and sustainable practices with ecosystem services, which is the basis for evaluating the efficacy of schemes such as PES and similar, and for promoting their application.

Concluding remarks

The book provides a framework for policy evaluation of impacts due to climate and land-use change on tropical forests in Central America, with a view of integrating ecological and economic science. Information about the cost of policy inaction is of outmost importance for decision-making about future development paths, given the crucial role of tropical forests in the conservation of biodiversity and for rural communities which depend strongly on their products and services.

The translation of ecological impacts into economic terms is one of the key contributions of this book, within a broader vision aiming at integrating bio-physical processes and outputs, spatial disaggregation and monetary values.

Though acknowledging the existing imperfections of economic valuation in this context, with unresolved issues such as those linked to nonlinearity, thresholds effects and irreversibility, still economic valuation can play a role in the policy context. This will depend on the capability of conveying transparent estimates embracing future scenarios of changes in the valuation, while providing appropriate context for their use.

The outcomes presented in this book should help decision-makers in designing appropriate conservation policies, contributing to a more sustainable management of the resources, while reducing ecological and human vulnerability.

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Part I

Central American tropical forests, ecosystem services and human wellbeing

1 An overview of forest biomes and ecoregions of Central America

Lenin Corrales, Claudia Bouroncle and Juan Carlos Zamora

Introduction

Central America is situated between the Nearctic and the Neotropical biogeographic realms, which have produced conditions for relatively high levels of biological diversity, with important ecoregions, ecosystems, endemism, and species richness. The Central America region comprises El Salvador, Belize, Honduras, Guatemala, Nicaragua, Costa Rica, and Panama, and it has been identified as part of one of the planet's richest and most highly threatened biodiversity regions, making it one of the world's top 25 biodiversity conservation hot spots. Considering species diversity and endemism, Central America is the world's second ranking priority hot spot in terms of plant and animal endemism, and fifth among all hot spots (Myers *et al.* 2000; Conservation International 2011).

This chapter describes the terrestrial ecoregions of Central America and its nested biomes based on the analysis framework proposed by Dinerstein *et al.* (1995) for setting priorities of conservation. Ecoregions provide a conceptual framework for the identification of representative habitats and serves as a tool to compare areas with different biodiversity features, status of their natural habitats and degree of protection (Olson *et al.* 2001). The description of their biotic characteristics, landscape context, main threats to loss of biodiversity and degradation caused by human activities and conservation status is based on the available literature but also on spatial analysis at regional scale.

Biomes and ecoregions description

The Selva Maya in Guatemala and Belize and the Darien forest in Panama are the northern and southern ends of Central American ecoregions. The isthmus (80–500 km wide) is bordered by the Pacific Ocean and the Caribbean Sea and divided by a mountain range reaching higher elevations in Guatemala and Costa Rica (4220 m on Mt. Tajumulco and 3820 m on Chirripó, respectively; DeClerck *et al.* 2010). The distribution and biophysical characteristics of four terrestrial tropical biomes (major terrestrial habitats; Olson *et al.* 2001; Figure 1.1) and 19

Box 1.1 Forest types and land use change in Central America

The forest cover in Central America has been changing since pre-Columbian times (Piperno 2006). After the Spanish conquest, the conversion of the Pacific dry broadleaf forests started with the introduction of cattle production in 1521 (Heckandon 2003). Studies for Costa Rica illustrate the role played by the agroecological conditions in the spatial patterns of deforestation and fragmentation common in the region, clearly showing two processes: the progression of vegetation loss from dry and low areas and towards wetter and steeper areas and the fragmentation of forest remnants associated with the expansion of annual crops and pastures (Finegan and Bouroncle 2008). The expansion of the cattle production and agriculture reached the wet Atlantic lowlands in the last decades of the XIX century, such that virtually no place of the Central American isthmus remains unaffected (Table 1.1). Other drivers of human -induced biodiversity loss that are increasing its significance, are over-exploitation of natural resources, water pollution (Programa Estado de la Nación en Desarrollo Humano Sostenible 2001), introduction of non-native species (Harvey *et al.* 2005a) and climate change (e.g. CATHALAC 2008; Clark *et al.* 2003; Feeley *et al.* 2007; Pounds *et al.* 2006; Whitfield *et al.* 2007).

Today the most important agricultural commodities in the region are coffee, sugarcane, oil palm, bananas and other tropical fruits, which accounted for 14 per cent of the total goods exported from the Central American countries in 2012 (ECLAC 2014). However, in terms of total land area occupied, the dominant agricultural land use is pasture for cattle raising which accounts for 2 to 46 per cent of the land use depending on the country, the importance of other agricultural activities varies across countries (FAO 2014). Basic grains (i.e. rice, corn, beans, sorghum) are the second-most common agricultural land use in El Salvador, Guatemala, Nicaragua and Honduras, whereas in Costa Rica, Belize and Panama, no single agricultural crop dominates (Harvey *et al.* 2005a). Although the economic importance of agriculture has declined in recent decades in all Central American countries except Nicaragua, it still remains an important economic activity, contributing to an estimated 7 to 36 per cent of the total gross domestic product per country (Programa Estado de la Nación en Desarrollo Humano Sostenible 2011).

Although highly deforested and fragmented, many of the landscapes still retain some on-farm tree cover, in the form of small (and often isolated) forest fragments, strips of riparian forest, dispersed trees in pastures, fallows and/or live fences. This on-farm tree cover is important for both farm productivity (providing shade and forage for cattle, while providing timber and firewood to farmers), and for biodiversity conservation at different scales (e.g. providing important habitat and resources to wildlife, as well as serving as corridors or stepping stones that facilitate animal movement) (Harvey *et al.* 2005b, 2008).

Table 1.1 Forest area changes in Central America between 1990 and 2010

Country	*Forest area (1,000 ha)*				*Annual change rate*					
	1990	*2000*	*2005*	*2010*	*1990–2000*		*2000–2005*		*2005–2010*	
					1,000 ha/yr	%	*1,000 ha/yr*	%	*1,000 ha/yr*	%
Belize	1,586	1,489	1,441	1,393	–10	–0.63	–10	–0.65	–10	–0.68
Costa Rica	2,564	2,376	2,491	2,605	–19	–0.76	23	0.95	23	0.90
El Salvador	377	332	309	287	–5	–1.26	–5	–1.43	–4	–1.47
Guatemala	4,748	4,208	3,938	3,657	–54	–1.20	–54	–1.32	–56	–1.47
Honduras	8,136	6,392	5,792	5,192	–174	–2.38	–120	–1.95	–120	–2.16
Nicaragua	4,514	3,814	3,464	3,114	–70	–1.67	–70	–1.91	–70	–2.11
Panama	3,792	3,369	3,310	3,251	–42	–1.18	–12	–0.35	–12	–0.36
Central America	25,717	21,980	20,745	19,499	–374	–1.56	–247	–1.15	–249	–1.23

Source: Adapted from FAO (2010).

terrestrial ecoregions (distinct assemblage of natural communities and species; Olson *et al.* 2001; Figures 1.2–1.5) are related to the physiography and precipitation distribution in the region, according to the Holdridge life zones (1979) that were used to delineate biomes and ecoregions in Central America (Dinerstein *et al.* 1995). Considering their community types and potential distribution, the most extensive are the moist broadleaf forests (55 per cent of the terrestrial area of the region), the coniferous forests (24 per cent), and the dry broadleaf forests (20 per cent), one per cent is covered by xeric shrublands. The montane ecoregions of the moist broadleaf forest and the coniferous forests biomes contains tropical montane cloud forests, ecosystems with characteristic structure and flora that occur in an altitudinal range with a seasonal or persistent cloud cover that receive higher precipitation and lower evapotranspiration (Brown and Kappelle 2001; Sáenz and Mulligan 2013), and in consequence are critical for the provision of hydric ecological services.

Each ecoregions average altitude, average annual rainfall, and dry months (Table 1.2), calculated from spatial databases, show the relationship between these biophysical characteristics of the ecoregions and species number of some vertebrate taxa. The values of indicators of ecoregions threats, ecologic integrity and conservation status of the Central American ecoregions (Table 1.3), also calculated from spatial databases, illustrate the different degrees of land use change on the ecoregions for the year 2000.

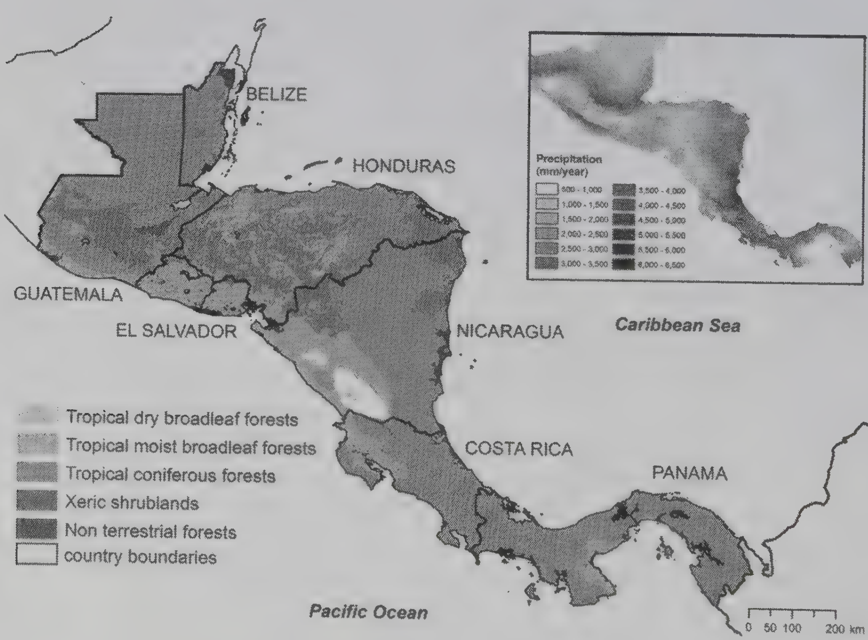

Figure 1.1 Biomes of Central America

Source: based on Olson *et al.* (2001) and Jarvis *et al.* (2008).

Table 1.2 Biophisical characterization and species richness of ecoregions in Central America

Biome / Ecoregion	*Size (km²)*	*Biophysical characteristics*			*Total number of species (percentage of threatened species)*				
		Mean altitude (masl)	*Mean annual precipitation (mm)*	*Months with mean precipitation < 100 mm*	*Amphibians*	*Birds*	*Mammals*	*Reptiles*	*Total*
Tropical moist broadleaf forests	260,575								
Lowlands ecoregions									
Central American Atlantic moist forests	90,513	295	2,333	3.3	38 (11)	429 (1)	172 (2)	143 (3)	782 (2)
Chocó–Darién moist forests	10,294	163	2,514	3.2	138 (15)	600 (3)	215 (5)	200 (1)	1,153 (4)
Costa Rican seasonal moist forests	7,566	585	1,968	5.1	40 (10)	373 (1)	186 (4)	99 (1)	698 (2)
Isthmian–Atlantic moist forests	45,431	181	3,187	1.9	118 (22)	518 (1)	217 (5)	168 (1)	1,021 (4)
Isthmian–Pacific moist forests	42,965	322	2,686	3.8	99 (18)	407 (1)	190 (4)	143 (1)	839 (4)
Petén–Veracruz moist forests	2,778	255	2,245	4.2	103 (21)	468 (1)	191 (6)	226 (4)	988 (6)
Montane ecoregions									
Central American montane forests	17,828	1,648	1,774	5.3	73 (53)	707 (< 1)	191 (3)	111 (0)	1,082 (4)
Chiapas montane forests	5,633	1,293	2,282	4.1	49 (41)	325 (0)	163 (2)	6 (0)	543 (4)
Eastern Panamanian montane forests	1,871	701	2,660	3.2	30 (7)	327 (2)	198 (5)	102 (0)	657 (3)
Sierra Madre de Chiapas moist forests	13,490	632	2,796	5.1	44 (36)	315 (0)	148 (3)	118 (2)	625 (4)
Talamancan montane forests	20,110	1,429	3,049	2.4	124 (42)	450 (2)	204 (5)	132 (0)	910 (8)

Table 1.2 Continued

Biome / Ecoregion	*Size (km²)*	*Biophysical characteristics*			*Total number of species (percentage of threatened species)*				
		Mean altitude (masl)	*Mean annual precipitation (mm)*	*Months with mean precipitation < 100 mm*	*Amphibians*	*Birds*	*Mammals*	*Reptiles*	*Total*
Tropical dry broadleaf forests	93,113								
Central American dry forests	74,632	243	1,646	5.8	36 (8)	330 (0)	195 (4)	99 (3)	660 (2)
Chiapas Depression dry forests	13,415	758	1,013	8.0	33 (24)	188 (1)	160 (2)	106 (2)	487 (3)
Panamanian dry forests	5,086	81	1,565	5.0	22 (5)	273 (< 1)	165 (2)	59 (2)	519 (1)
Tropical coniferous forests	117,755								
Central American pine–oak forests	97,494	1,123	1,485	5.8	107 (47)	349 (1)	203 (5)	194 (1)	853 (8)
Miskito pine forests	17,412	38	2,694	2.8		240 (< 1)	128 (2)		368 (1)
Belizean pine forests	2,849	150	2,074	3.4	14 (0)	272 (1)	122 (2)	39 (0)	447 (1)
Xeric shrublands	2,200								
Motagua Valley thornscrub	2,200	645	1,177	7.1	23 (0)	115 (0)	138 (1)	88 (2)	364 (1)

Source: Prepared from Olson *et al.* (2001), Hijmans *et al.* (2005), and WWF (2006).

Notes: All indicators were calculated considering the extent of the ecoregions within the countries of Central America. The reference period of climatic variables is 1950–2000.

Table 1.3 Indicators of ecoregions threats, ecologic integrity and conservation status in Central America (year 2000)

Biome	*Ecoregion*	*Average annual rate of deforestation (1990–2000)*	*Roads density (m/km^2)*	*Fires density (hot points/km^2)*	*Largest natural fragment (km^2)*	*Remaining original cover (%)*	*Territory under protection (%)*	*Conservation Risk index*
Tropical moist broadleaf forests (lowlands)	Central American Atlantic moist forests	0.94	47	0.08	14,629	53	20	2.4
	Chocó–Darién moist forests	0.05	4	0.00	10,063	98	10	0.2
	Costa Rican seasonal moist forests	1.50	436	0.06	228	32	12	5.7
	Isthmian–Atlantic moist forests	0.94	119	0.02	5,206	55	16	2.9
	Isthmian–Pacific moist forests	1.05	206	0.06	1,880	37	5	4.9
	Petén–Veracruz moist forests	Not rated	320	0.11	647	60	52	0.8
Tropical moist broadleaf forests (montane)	Central American montane forests	1.05	241	0.12	927	65	47	0.7
	Chiapas montane forests	1.16	128	0.16	828	79	0	100.0
	Eastern Panamanian montane forests	0.10	12	0.01	408	96	63	0.1
	Sierra Madre de Chiapas moist forest	1.16	645	0.24	142	14	1	86.0
	Talamancan montane forests	0.41	100	0.01	8,794	77	66	0.3
Tropical dry broadleaf forests	Central American dry forests	0.62	454	0.18	326	22	2	39.0
	Chiapas Depression dry forests	0.30	224	0.19	516	45	0	100.0
	Panamanian dry forests	0.83	433	0.10	5	5	7	13.6
Tropical coniferous forests	Central American pine–oak forests	1.16	225	0.13	3,019	47	8	6.6
	Miskito pine forests	0.41	70	0.20	9,162	96	4	1.0
	Belizean pine forests	Not rated	No data	No data	No data	71	32	0.9
Xeric shrublands	Motagua Valley thornscrub	0.41	482	0.11	786	63	22	1.7

Source: Prepared from Olson *et al.* (2001), Vreugdenhil *et al.* (2002), Davies *et al.* (2009), IUCN & UNEP–WCMC (2014).

Note: The Conservation Risk Index (CRI; Hoekstra *et al.* 2004) considers the relationship between the proportion of area converted and the proportion of area protected.

Tropical moist broadleaf forests

This biome is by far the largest, considering it potential distribution. It is also the one that contains the highest number of ecoregions. In Central America, it extends in a continuous strip from Guatemala to Southern Panama, along the Atlantic Coast in lowlands and montane areas (Figure 1.1) with low annual temperature variability and high rainfall (2000–3200 mm annual precipitation). The forests is dominated by semi-evergreen and evergreen deciduous tree species arranged in up to five strata, and is the highest in species diversity in any terrestrial biome in the world. According to their altitudinal distribution, the ecoregions of this biome in Central America can be grouped in lowlands and montane ecoregions. The first ones are distributed in continuous strips (Figure 1.2), while the montane ecoregions are naturally fragmented (Figure 1.3).

In general, the ecoregions of this biome are in very different states of ecological integrity, reflecting the intensity of human productive activities and land occupation. The humid forests of the Chocó Darien ecoregion have the best ecological integrity (98 per cent of remaining original cover) in Central America while the Costa Rican seasonal moist forests have the worst (32 per cent of remaining original cover) and the higher natural habitat loss rate.

The Central American Atlantic moist forest ecoregion covers the lowland Atlantic slopes of northern Central America and is the second largest ecoregion in the isthmus (Figure 1.2). It is in the middle range of species richness in Central

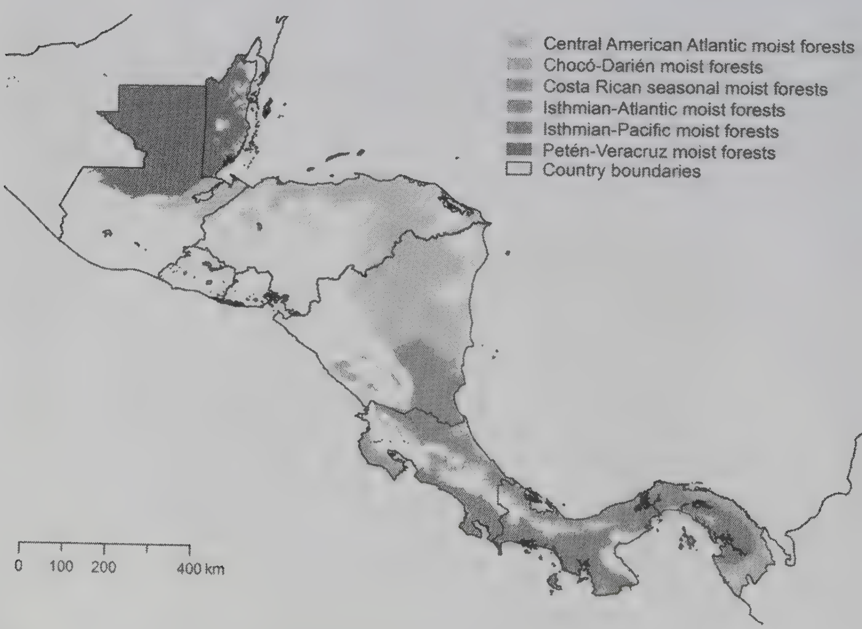

Figure 1.2 Tropical moist broadleaf forests (lowland) ecoregions of Central America
Source: based on Olson *et al.* (2001).

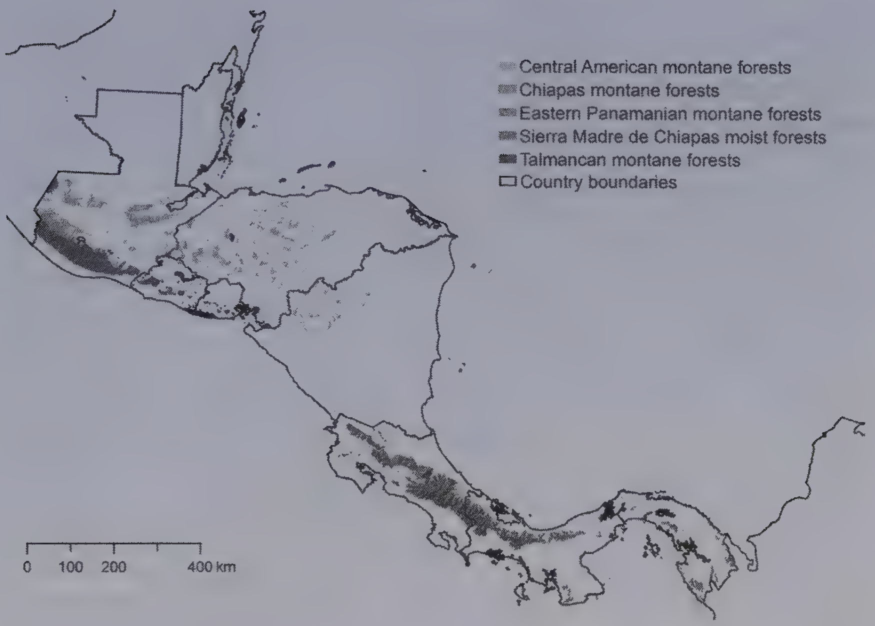

Figure 1.3 Tropical moist broadleaf forests (montane) ecoregions of Central America
Source: based on Olson *et al.* (2001).

America (Table 1.2), with characteristic tropical evergreen forest with canopy trees reaching 50 m in height (Powell, Palminteri, and Schipper 2001a).

The main drivers of habitat conversion are the agricultural and infrastructure expansion, wood extraction and other indirect causes (Geist and Lambin 2001). About half of the original habitat remains and the ecoregion conserves natural habitat fragments bigger than 100 km^2 (Table 1.3), a size than could maintain viable populations of large vertebrates (Dinerstein *et al.* 1995; Powell, Barborak, and Rodriguez 2000); the low density of habitat fragments is related to a low degree of fragmentation, fires and roads density are not critical for this ecoregion (Table 1.3).

The Chocó-Darién moist forests ecoregion covers lowlands in eastern Panama in Central America (Figure 1.2) and the entire Pacific Coast of Colombia. This is one of the few places in the Neotropics with pluvial rainforest (D'Ambrosio 2001). This ecoregion is the richest of Central America, considering the total species of amphibians, birds, mammals and reptiles (Table 1.2); it is also considered one of the most biodiverse lowland areas in the world, with exceptional abundance and endemism of plants, birds, amphibians, and butterflies. The ecoregion has the best ecological condition in the region since almost the entire original habitat remains in large forest blocks with minimal fragmentation and high connectivity. The proportion of natural habitat converted is very low (Table 1.3) due to its low population density and Panamanian regulations on livestock expansion activities.

The Costa Rican seasonal moist forests ecoregion lies between the crests of Costa Rica's Central Volcanic Cordillera and the Pacific Ocean (Figure 1.2). It is naturally fragmented and covers the Pacific slopes, most of the Nicoya peninsula and small areas in Nicaragua. These forests link mountain tops and the Atlantic slopes, an aspect that facilitates the migration of many species (Imbach *et al.* 2013). Deciduous trees that lose their leaves during the dry season dominate these forests (Powell, Palminteri, and Schipper 2001b). Only one third or the original habitat remains and is very fragmented (as can be deduced from the high density of fragments remaining habitat, the highest between the Central American ecoregions) with fragments smaller than 50 km^2 (the minimal habitat size for middle-size felines; Stiles and Skutch 1989). However, these small fragments can be valuable to preserve communities and representative species, as well as stepping stones for ecological connectivity. This ecoregion has the higher loss rate of original habitat, two of the main causes of habitat loss have been the agricultural expansion and development of major cities in Costa Rica across what is called the Central Valley (see the high level of roads density in Table 1.3).

The Isthmian-Atlantic moist forest ecoregion covers lowlands from southern Nicaragua and along the eastern Caribbean Coast of Panama (Figure 1.2) with high rates of precipitation. The forests have a complex structure that includes palms in the medium canopy strata, and a very rich epiphyte flora. This ecoregion is the second richest of Central America, considering the total species of amphibians, birds, mammals and reptiles (Table 1.2). Seasonal swamp forests occur in the lowest and flattest areas in Nicaragua and northern Costa Rica (Powell, Palminteri, and Schipper 2001c). The main driver of habitat conversion is the agricultural expansion, high road density is concentrated in Costa Rican and Panama areas (Table 1.3), so it is expected that the degree of fragmentation differs along the ecoregion.

The Isthmian-Pacific moist forest ecoregion includes the slopes from southern Costa Rica and eastern Panama (Figure 1.2). Its location in the Intertropical Convergence Zone makes it one of the wettest in the region, as the moisture-laden winds from both the north and south collide causing a long rainy season and a short dry season. Many species of South American affinity find here its northern distribution limit. Highly diverse in plant and animal species, where many new species of angiosperms from the Osa peninsula in Costa Rica are continuously reported. Several characteristic plant species of the area are in danger of extinction since much of the ecoregion has been destroyed by human settlements, pastures, coffee, pineapple, and oil palm plantations (see the high annual rate of deforestation between 1990 and 2000 in Table 1.3) and is scarcely protected (only 5 per cent of its area; Table 1.3).

The Petén-Veracruz moist forests ecoregion lies in coastal lowlands along southern Mexico, Belize and northern Guatemala, between the Gulf of Mexico and the Septentrional Sierra Madre de Chiapas (Figure 1.2). It consists of a matrix of wetlands, riparian habitats, and moist forests. Species richness is high, although endemism is relatively low. Almost 60 per cent of the original habitat remains, some fragments are larger than 50 km^2, enough for medium-sized cats, but probably

with a high loss rate of original habitat due the fires and increased road density (Table 1.3).

The Central American montane forest ecoregion is made up of forest patches on the isolated tops and slopes of the highest mountains of Central America, from southern Mexico into northern Nicaragua (Figure 1.3); with a temperate climate and high precipitation. It is in the higher range of species richness in Central America (Table 1.2). Subsistence farming has altered the lower elevations of this ecoregion (Powell and Palminteri 2001), a process which is causing an accelerated loss of habitat. Most of the higher altitude habitats are formally protected in the different countries (Table 1.3), but not always put into practice.

Most of the Chiapas montane forests ecoregion is located in southern Mexico, but a small portion is included in northern Guatemala (Figure 1.3). This ecoregion has a temperate climate and high rainfall, the climate along his steep slopes is extremely humid, and mist is almost always present. They have a natural patchy distribution bounded by lowland moist forests to the east and the pine-oak ecoregion to the west. The richness in avifaunal species is considerably high (considering the relative small size of the ecoregion) (Table 1.2). The remaining original cover is being loss by the high rate of deforestation and fires (Table 1.3).

The Eastern Panamanian montane forests ecoregion, the smaller ecoregion in Central America, is located in western Panama, surrounded by the Chocó-Darien moist forest and the Isthmian-Atlantic moist forests in the lowlands (Figure 1.3). This ecoregion includes various vegetation types (marshes, swamp forests, semi deciduous tropical forests, premontane wet forests, cloud and elfin forests), and has high species endemism (Powell *et al.* 2001a). The largest natural fragment, smaller than 50 km^2, can be attributed to the natural patchy distribution of these montane forests. The extension of the Pan-American Highway has led to a slow rate of natural habitat loss between 1990 and 2000 (Table 1.3). Protected areas (Darien National Park, forests and indigenous reserves) cover 90 per cent of the ecoregion (Table 1.3).

The Sierra Madre de Chiapas moist forest ecoregion extends along the Sierra Madre of Chiapas Mountains through Mexico, Guatemala, and El Salvador (Figure 1.3). All the ecoregion is considered an Important Bird Area (according to Devenish *et al.* 2009), with over three hundred species of avifauna, and also a center of endemism for salamanders and butterflies (Valero, Schipper, and Allnutt 2001a). About 14 per cent of the original habitat remains and is highly fragmented. It has the highest fires and roads density among the moist broadleaf ecoregions in Central America (Table 1.2), and with the Chiapas montane forest and Central American pine-oak forests, has had the higher rate of natural habitat loss in the region (Table 1.3). Only 7 per cent of the ecoregion is under protected areas.

Most of the Talamancan montane forests ecoregion is located in the highlands of Costa Rica (Figure 1.3), the southern limit of distribution reach western Panama and has high species richness (Table 1.2). It is the most important area of endemism of terrestrial vertebrates in Costa Rica, concentrating about 80 per cent of the endemic species of the country, highlighting a rich herpetofauna. Over 30

per cent of the ecoregion's flora is endemic to this area, the cloud forests of the upper reaches of the mountains are highly endemic in flora species, especially in ferns and orchids (Powell *et al.* 2001b; SINAC and INBio 1999). Almost 77 per cent of its original forest cover remains intact without significant fragmentation, fire and roads density and the rate of deforestation (Table 1.3) are not among the highest in the region as these are more severe in the mid elevations zones. More than 60 per cent of the ecoregion is under protected areas (Table 1.3), more of them national parks (IUCN Category II). The clearing of forests for agriculture development, cattle pastures, timber harvesting and human settlements has altered the unprotected habitat.

Tropical dry broadleaf forests

This biome extended originally in a continuous strip from the Pacific Coast of southern Chiapas (Mexico) to northwestern Costa Rica with a much smaller strip around the Bay of Panama (Figure 1.1) and multiple fragments scattered in interior lowland areas surrounded mostly by coniferous forests. This biome occurs in lowland and premontane areas with low annual temperature variability, low rainfall (< 2000 mm annually) and long dry seasons (5–8 months). The forests of this biome are generally composed of two strata, a higher story of deciduous trees and a lower story that usually includes evergreen species. Species tend to have wider ranges than moist forest species, although some species display highly restricted ranges. Beta diversity is high but typically lower than adjacent moist forests.

Tropical dry forests have suffered much more pressure from anthropogenic activities than tropical rainforests, less than 1 per cent of the original area of dry forest remains in Central America (Janzen 1988). The factors that underlie the greater human impact on dry forests include the favorable climate for both humans and cattle, easier burning for shifting cultivation practices, and lower leaching of soil nutrients from agroecosystems, along with greater soil resistance to compaction (Maass 1995). Today, some efforts of conservation (local protected areas and reforestation) and the secondary succession in abandoned pastures show different options to regrow forests (Griscom and Ashton 2011). The dry forests of the Chiapas Depression ecoregion have the best condition in Central America context while the Panamanian dry forests have had the higher natural habitat loss.

The Central American dry forest ecoregion, which stretches in lowland and premontane areas along the Pacific Coast (Figure 1.4), corresponds to a tropical habitat that has a prolonged dry season. It is home to important plant and animal species, as well as a significant degree of endemism (Andraka 2001). Only 22 per cent of the original habitat remains, the larger by far in this biome, but it shows the highest fragment density in Central America with the largest fragments smaller than 50 km^2. Most of the hydrological basins are contaminated and experience frequent droughts. Cattle, agricultural expansion, burning, as well as water pollution, put strong pressures in the scarcely protected remnants (Table 1.3).

The Chiapas Depression dry forests ecoregion lies on flat lands between the Chiapas plateau and the Sierra Madre of Chiapas (Figure 1.4), and is formed by

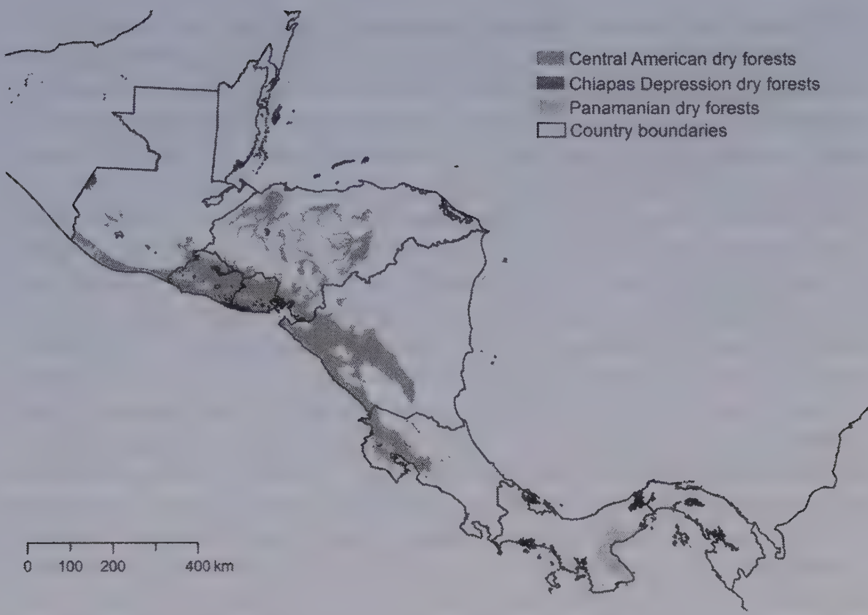

Figure 1.4 Tropical dry broadleaf forests ecoregions of Central America
Source: based on Olson *et al.* (2001).

the convergence of two biogeographically important routes: one from the Gulf of Mexico and other from the Pacific coast. The ecoregion contains species that have migrated to the dry forests through those corridors and has a very high degree of endemism (Valero, Schipper and Allnutt 2001b). Only 45 per cent of the original cover remains and has one of the highest fragments density in Central America although still retains fragments larger than 50 km^2. Cattle grazing, along with intensive logging for agricultural purposes, human settlement and fires, have contributed to the majority of habitat destruction, and continue to threaten the poorly protected remnants (Table 1.3).

The Panamanian dry forests ecoregion is on the Pacific coast, around the Gulf of Panama, it and it is disconnected from other dry forests (Figure 1.4). Its vegetation has been significantly altered and only 5 per cent of the potential distribution remains in small fragments. Cattle ranching, fires, and hunting remain strong pressures, although virtually all remnants are within protected areas (Table 1.3).

Tropical coniferous forests

The forests of this biome are distributed along the upper two-thirds of Central America along the northern Caribbean coast, the lowlands of Honduras and Nicaragua and the along the mountain ranges that comes from the Sierra Madre de Chiapas until northern Nicaragua (Figure 1.1).

The areas have variable levels of precipitation and moderate variability in temperature. The most outstanding characteristic of the biome, one that marks the southern limit of boreal floristic influence in the New World, is the richness of the *Pinus* genus (and in some areas its assemblage with *Quercus* spp.), which is adapted to the variable climatic conditions and natural fires. The biome status is threatened by agriculture expansion, logging, forest fires, pests and firewood use.

The Central American pine-oak forests ecoregion extends from the Sierra Madre de Chiapas (southern Mexico) until northern Nicaragua (Figure 1.5). The dominant species belong to the *Pinus* and *Quercus* genera; pine and oaks forests often form intricate mosaics and complex successional interactions, also often present as mixed forests between *Pinus* and islands of broad-leaved cloud forests at higher altitudes (Rzedowski 2006).

This ecoregion is in the middle range of species richness (Table 1.2), and contains some endemic flora—including tree species (Gómez-Pompa and Dirzo cited in Powell *et al.* 2001c), the greatest diversity of pine species occurs in areas where the topography is higher and presents abrupt changes (Perry, Graham, and Richardson 1993). The ecoregion contains also an Endemic Bird Area (Devenish *et al.* 2009), 16 species of endemic amphibians and 24 species of endemic reptiles (Flores-Villela cited in Powell *et al.* 2001c). Only 47 per cent of the original cover remains and is highly fragmented (Table 1.3) due to high population density, dominated by subsistence farming practices. The loss of natural habitat between 1990 and 2000 was one of the highest in Central America, some habitat remains

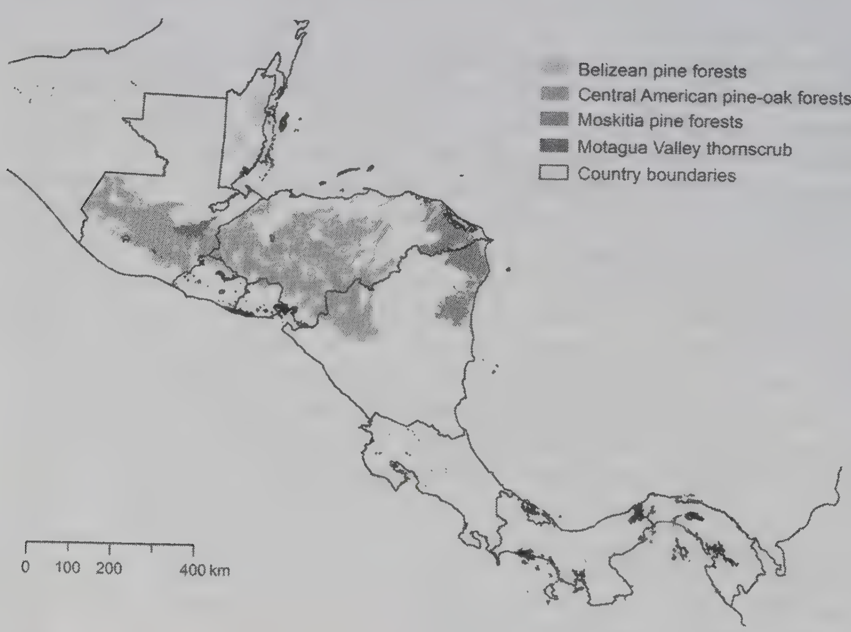

Figure 1.5 Tropical coniferous forests and xeric shrublands ecoregions of Central America
Source: based on Olson *et al.* (2001).

almost intact in Honduras but the few protected areas in the ecoregion (Table 1.3) are primarily designed to protect cloud forests, and do not include large tracts of pine-oak forests (Powell *et al.* 2001c).

The Miskito pine forests ecoregion is restricted to the Caribbean coast and lowlands of Honduras and Nicaragua (Figure 1.5). After the Motagua Valley thornscrub, this ecoregion is the less rich ecoregion in species of macrofauna of Central America (no amphibians of reptiles are reported here, Table 1.2), but the high variety of ecosystems, some of them endemic (Meyrat and Gretzinger 2002) are biodiversity features of interest.

Other formations, additional to the predominant pine savannah, are the palm-savanna, flooded grasslands, gallery forests, and swamps. Periodic low-intensity fires help maintain open savanna and hence the dominance of pine (*Pinus caribaea* var. *hondurensis*; Stevens *et al.* 2001).

The ecoregion conserves more than 95 per cent of its original cover with no evident fragmentation (Table 1.2), but some forests show low rates of regeneration of *Pinus* species and hardwoods, probably a result of indiscriminate logging, cattle, and frequent intentional fires (Vreugdenhil *et al.* 2002; Delgado *et al.* 2007). Despite this ecoregion has one of the highest fires densities in Central America, the average annual rate of deforestation between 1990 and 2000 was moderate (0.41), probably because of low population density of the area. Pressures such as indiscriminate logging and the expansion of monocultures can increase the degradation of the natural cover, which has less than 20 per cent is protected (Table 1.3).

The Belizean pine forest ecoregion lies in the Central America's northwestern Caribbean coast (Figure 1.5). Similar to other pine forests, the Belizean pine forests are adapted to nutrient-poor soils and a regimen of periodic low-intensity fires. The forest configuration of this ecoregion, where the dominant tree species is also *Pinus caribaea* var. *hondurensis*, is related to an elevation gradient between the coastal plains and the Maya Mountains. The forests of the montane areas are taller and denser and occur in relatively large fragments, while those in the lowlands are lower, sparse and more fragmented. It is the latter which are most threatened by logging and agricultural expansion (Andraka, Locklin, and Schipper 2001).

Xeric shrublands

This biome is restricted to one ecoregion in the north of Central America (in the Guatemala–Honduras border). The Motagua Valley thornscrub ecoregion is a valley surrounded by mountains that reach up to 3,000 m (Figures 1.1 and 1.5), where mountains facing the northeast trade winds produces a rain shadow effect causing arid conditions in the valley, high temperatures (> 41°C), and a long dry season (Vreugdenhil *et al.* 2002), features that makes this ecoregion very different from the neighboring ones. For example, this ecoregion is the less rich in species of macrofauna of Central America (Table 1.2) in high contrast with the adjacent cloud forests (of the Mesoamerican pine-oak forest ecoregion) in the North. The ecoregion is dominated by two types of vegetation, the thorn

scrub and the dry forest, but has a high floristic diversity of tree communities mostly as riparian habitats (Castañeda 1995; Corrales 1997). The valley has been almost entirely converted to irrigated croplands while in the slopes subsistence agriculture prevails (Nájera 2006), a consistent situation with the high fire and road density of this ecoregion. Current land use trends threat to eliminate the remnants of the native biodiversity since only 14 per cent of the ecoregion is protected (Table 1.3).

Final remarks

Available information, focused in particular areas and special taxa such as birds, mammals, reptiles or insects, shows that the Central America ecoregions have a large gradient of species diversity, the ones situated in southern Central America—the wettest and lowest sector—have greater total species richness of macro-fauna while dry and coniferous forests have higher rates of endemism. A landscape approach in future studies with focus on species distribution could be useful to know biodiversity distribution patterns within each ecoregion.

The forests of all ecoregions provide a range of marketable and non-marketable ecosystem services, such as timber, firewood, shade, water and cultural services, with a gradient on the offer from each ecoregion. The most obvious is that the montane forests have a critical role in hydric ecosystem services (Sáenz and Mulligan 2013) relative to other ecoregions. Also the storage and production of organic matter have a relationship with climate, with peak values in the humid forests and lower values in wetter and dryer forests (Brown and Lugo 1982).

Land use change is the main cause of natural habitats loss in Central America, affecting ecoregions and ecosystem services provision. In the dry forest ecoregions on the Pacific slope of the region, with favorable climate and soils for agriculture, cattle, and urban area development, land use change processes have a longer history and cover larger areas, whereas in humid forest ecoregions with less favorable conditions, are less affected. Ecoregions in the north of Central America, with higher density of human population, are also more affected than its southern counterparts.

Climate change will also affect the natural habitats of the ecoregions, and ecosystem services they provide. Under future greenhouse emission scenarios, Central America is the region with largest expected changes in climate among tropical areas (Giorgi 2006; Rauscher *et al.* 2008). Some changes such as transition between vegetation types are expected. In an regional analysis, Imbach *et al.* (2012) estimated, for the 2070–2100 period, ecosystems could suffer a reduction in leaf area index across 77–89 per cent of the region, resulting in a reduction of trees canopy cover and a transitions of tropical rainforest to seasonal or dry forests, and a potential increase in bushlands and pastures. In a specific study in Costa Rica, Kannan and James (2009) show that there will be significant change in precipitation quantity and variability as well as a rise in the height of cloud formation in the Pacific will have severe consequences on montane forests and their hydrological services. The analysis of the impact of

land use change in provision of ecosystem services in Southern Mexico, that includes tropical seasonal forests, pine-oak forests and montane forests, shows changes in species composition and diversity and water quality declination (Martínez *et al.* 2009).

Natural habitat condition across ecoregions not only affects the ecosystem services they provide, but also the ability to adapt to climate change. It is expected that migration will be a basic response of vegetation to climate change (since the rate of change in climate will not allow for genetic adaptation). Habitat loss and fragmentation may affect the capacity of plants to migrate by reducing seed sources and the availability of dispersal agents of different ecological groups, which combined with species dispersal rates could constrain species distribution across the region under climate change (Imbach *et al.* 2013).

Ecoregions in Central America also differ in the proportion of its territory under protection, which varies between 1 per cent for the Sierra Madre de Chiapas moist forests and 66 per cent for the Talamancan montane forests. This

Box 1.2 Policy implications for ecosystem services conservation

- Beyond the biophysical marked differences between the ecoregions of Central America, their ecosystems provide a range of ecosystem services that contribute to food security through climate regulation, soil fertility, pollination and water supply for irrigation.
- The potential changes in Central America climate under future greenhouse emission scenarios, makes this region the one with largest expected changes in climate among tropical areas. So climate change will affect the natural habitats of the ecoregions, and the ecosystem services they provide.
- The resilience of these habitats varies in the region, both for their intrinsic characteristics such as different degrees of degradation by changing the land use and other human processes.
- There is growing consensus that protected areas only, are insufficient to conserve the rich Central American biodiversity and the ecosystem services it provides. Considering the ecosystem services in the decision-making, beyond the conservation sector, will be of vital importance for the adaptation to climate change in society.
- The restoration of natural habitats could be of particular importance. It will not only contribute to carbon sequestration (a priority of the political agenda in the region in the past decade) or improve the resilience of forest systems to address climate change, but also the maintenance of livelihoods of people living in rural areas.
- Considering ecosystem services in planning processes contribute to disaster risk reduction and in addition to adaptation to climate change.

shows that protected areas systems in the region have a large gradient of representativeness of ecoregions, showing a bias towards the tropical moist broadleaf forest in mountain areas. Some of the protected areas systems were established when most of the lowlands and drier habitats were lost. Also, more effort was done to ensure the provision of hydric services, protecting mountain forests. According to the Conservation Risk Index (CRI; Hoekstra *et al.* 1994), the ecoregions with higher relative risk of biodiversity loss are the Sierra Madre de Chiapas moist forests, followed by the Central American dry forests and the Panamanian dry forests. Ecoregions with the lowest values of risk are the Eastern Panamanian montane forests, the Chocó-Darién moist forests and the Talamancan montane forests.

Although protected areas are the keystone to all conservation efforts, they are often too small, too isolated and heavily impacted by human activities. They also do not protect all of existing ecosystems and habitat types across the region. Furthermore, most of the protected areas are surrounded by agricultural and pastoral landscapes, isolating animal populations and therefore reducing their long-term viability. Many protected areas are also threatened by human activities in adjacent lands, such as fires, pesticide contamination and hunting, among others. A recent analysis of conservation efforts in the region (DeClerck *et al.* 2010) emphasizes the need to continue strengthening the systems of protected areas and biological corridors, with strategies to promote diversity and connectivity in the agricultural matrix (e.g. Harvey *et al.* 2005b, 2006).

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2 Ecosystem services in tropical forests

Contribution to human wellbeing and implications for economic valuation

Aline Chiabai

Introduction

Tropical forests provide a wide range of services which sustain life, such as timber and non-wood forest products (e.g. wild food), climate regulation, soil retention, water regulation and supply, pollination, landscape aesthetics and genetic information yet to be discovered. They also provide habitats for a vast store of species. At the global scale, tropical forests are home to about half of all existing animal and plant species on the planet, although they cover just about 7 per cent of the land. These forests show in fact the highest species richness compared to other forest ecosystems (Gibson *et al.* 2011; Whitmore 1998; Gentry 1992).

In Central America there are approximately 22 million hectares of forests (FAO 2005), which adds up to 0.6 per cent of the global forest area. However, it is the region with the highest deforestation rate. As a matter of fact, in the period 2000–2005, around 285,000 hectares have been lost each year (FAO 2005), corresponding to a 1.3 per cent yearly loss in forest surface. Recent trends over the last decade indicate, however, a reduction in the deforestation rate in some tropical regions, especially in the Brazilian Amazon (IPCC 2014). Land-use change, deforestation and forest fires are considered principal causes for loss of forest cover in the Amazon, even more important than climate change (Settele *et al.* 2014). Nevertheless, the on-going degradation of forests throughout the Central American regions continues undebated with increasing fragmentation of human dominated landscapes on the Pacific slopes, and agricultural expansion into pristine Caribbean lowland forests. Conversion of pristine forests to other land uses has shown a significant decline in biodiversity levels, due to logging and agricultural activities which have expanded largely in the twentieth century. In addition to this loss, there is an increased vulnerability of species in the residual fragments of forest which might lead to their extinction. What is the impact of this fragmentation and the resultant loss in the functional capacity of these landscapes is a question that still needs to be answered. Degradation of large tracts on forests results in disruption of water supplies, deterioration of water quality (as the forest will no longer be able to filter and purify water), increased soil erosion (leading to

a higher frequency of landslides and the silting up of rivers), increased droughts in some places and flooding in others. Protecting the remaining tropical forest becomes a key factor in a context of global warming for many reasons. While the destruction of forests contributes to greenhouse gas emissions, healthy forest ecosystems provide services which can help in the climate change mitigation (e.g. through carbon sequestration) and adaptation (e.g. through reduction in soil erosion and protection of ground water supply; see Chapter 8 for a discussion on ecosystem-based adaptation). Despite the uncertainties on the impacts that climate change might have on forest ecosystems, the role of forests in this context is undeniable.

All in all, forest ecosystem services affect human wellbeing, including health, although the links might be complex and difficult to assess due to the multifaceted temporal and spatial interlaces. Some services are more directly linked with human health such as water supply, provision of wild food, recreational and aesthetic values. The latter, for example, benefit human health in many aspects. Forests create opportunities for active lifestyle through recreational activities and therefore play a role in reducing problems such as obesity and cardiovascular diseases, while encouraging other healthy behaviors such as the consumption of healthy food. Aesthetic values promote the reduction of stress and psychological tensions, thus contributing to mental health. Other benefits directly linked with human health and wellbeing include the possibility of reducing socio-economic health inequalities, improving air quality and microclimate, protecting water resources, obtaining pharmaceutical products and medicinal herbs, as well as improving community cohesion (Natural England 2009). Services such as soil retention or climate regulation are also linked with human health, although indirectly, as they provide a healthy environment for human beings and opportunities for climate change mitigation and adaptation.

Degradation of tropical forests triggered by land use changes can have, therefore, significant impacts on human health, not only for the loss of the above mentioned services, but also through other patterns, such as infectious disease outbreaks, or loss of habitats for animal species compelled to find other environments neighboring human infrastructures (ForHealth 2014; IUFRO undated). In these conditions, climate change would be an additional stressor which might exacerbate these impacts. Forests are certainly very receptive to changes in climate conditions. Climate factors such as temperature, solar radiation, rainfall and CO_2 influence strongly forest productivity and their functioning. On the other side, forests sequester large quantities of carbon from the atmosphere, they provide a cooling effect through evapotranspiration and clouds, thus contributing to control climate change. It is estimated that 40 per cent of carbon worldwide is stocked in tropical forests, which show also high sequestration rate of CO_2, while also contributing to more than half of gross primary production (Pan *et al.* 2011).

As for the expected impacts of climate change, the last IPCC assessment predicts (with medium confidence) that in the low mitigation scenarios there is a high risk of irreversible alterations in the functioning and composition of forests and other terrestrial ecosystems, with consequent modifications of vegetation types

and losses of forests especially in the Amazon and the Arctic (IPCC 2014). These changes are likely to be associated with a shift and alteration in habitats which will occur much more rapidly than the natural migration, triggering a change in plant and animal species, as well as increasing the risk of their extinction. In addition, these changes in forest composition, accompanied by degradation and deforestation, are expected to produce a great loss of carbon which will be released into the atmosphere, while the standing biomass will decrease due to the increased frequency of outbreaks, pests and forest fires. Forest fires will increase (with high confidence) due to the increased risk of droughts and because of land use changes, especially in moist tropical forests where there is higher vulnerability to droughts. Dry tropical forests in Central and South America, which have already seen a dramatic reduction due to human activities, are expected to face an increase in temperature and a decrease in precipitation patterns. These changes might increase the risk of degradation and replacement in dry tropical forests (Miles *et al.* 2006), though there is a low confidence in these predictions. Lastly, increased forest dieback in tropical regions are expected over the twenty-first century (with medium confidence), associated with higher temperatures and droughts, which can lead to further reductions in species diversity and spread of invasive species, while influencing the forest composition in terms of species and structure (IPCC 2014; Anderegg *et al.* 2012), with additional risks for carbon storage, timber production and water quality.

As a consequence of the mentioned changes, many ecosystem services provided by tropical forests will be under threat. Regarding biomass and soil carbon, although these are currently showing an increase in terrestrial ecosystems (including in the Amazon), they are highly vulnerable due to the rise in temperature, droughts and fire episodes expected for the twenty-first century (IPCC 2014). As for primary production, an increase is predicted by IPCC in terrestrial ecosystems compared with the pre-industrial era (low confidence; IPCC 2014). Trends in timber production are different according to the geographical area considered, though a general future trend of increase in forest production is expected with climate change (IPCC 2014; Kirilenko and Sedjo 2007). However, the effect cannot be clearly attributed to climate change alone, due to many other confounding factors in forest management. Increased forest dieback might decrease timber productivity in tropical regions.

Other important changes relate to habitat shifts and alteration of their quality, with risk of extinction and impacts on species abundance; increase in pest species and diseases which will affect forest productivity, and expected decline in pollination caused by a reduction in insect abundance, with effects on plant reproduction.

Still, water availability is expected to be one of the most affected services in forests, with higher risks of flooding and droughts (Settele *et al.* 2014; Kundzewicz *et al.* 2007).The effects of climate change on freshwater systems will exacerbate many forms of water pollution, and will impact on water system reliability and operating costs. Of all ecosystems, freshwater systems will have the highest proportion of species threatened with extinction due to climate change (MEA

2005), via the warming of water, flow alteration and loss of aquatic habitat (for instance, due to ice-jam flooding or decrease on river discharges).

Considering that Central and South America are highly dependent on water resources for the agricultural and hydropower sectors, significant impacts can be expected in this respect (IPCC 2014). In the case of hydropower, the loss of forest area might cause, in a first instance, a reduction of evapotranspiration, which would, on the one hand, increase water flows and affect positively energy production. On the other hand, however, it would influence the rainfall system which is sustained by the forest itself, reducing in this way the quantity of water flows as well as the water quality (e.g. sedimentation problems) for hydropower, so that these two effects have to be considered jointly when modelling the impacts (Stickler *et al.* 2013).

The expected changes in the provision of ecosystem services will affect human wellbeing and welfare in many respects. As already mentioned, there is an impact on human health (directly or indirectly affected), which can be subsequently translated into economic costs for the public health, in addition to other tangible and intangible costs for the population in terms of loss of productivity and leisure time, suffering and quality of life. More tangible impacts are expected for specific economic sectors, such as tourism, agriculture and hydropower production, for the high potential they represent in many regions located in tropical forests. The expected reduction in the provision of cultural services, water supply and regulation, will inevitably translate into economic losses which will be reflected in national economies. Further consequences are envisaged for forest-dependent groups and indigenous populations, though their monetization might pose some problems in the quantification.

Many of the impacts on ecosystem services mentioned above can be valued in monetary terms, as it is discussed later in the chapter, through market and non-market valuation methods. For some of them, the conversion into an economic unit becomes more difficult due to the nature of the service itself, as is the case for ecosystem services which are not directly traded in the market (e.g. water regulation, recreational activities, forest habitats). Despite these difficulties, the need for monetary values is undeniable in a policy context, where decisions have to be made about the use of natural resources and the adoption of appropriate mitigation and adaptation measures. Among the mitigation options, forest restoration and sustainable forest management can help in the general mitigation effort, while improving local economies and reducing poverty. Sustainable forest management (e.g. reduction of logging, prevention of fire, agro-forestry practices) can be also seen as a key measure for adaptation as it increases resilience of tropical forests. However, the decision to set up mitigation and adaptation plans relies necessarily on the knowledge about their economic costs and the flow of benefits they can generate in the short and long term. The economic valuation of ecosystem services represents, therefore, the cornerstone for applying cost-benefit analysis in decision-making processes.

The present chapter discusses the importance of ecosystem services provided by tropical forests, their utility for human wellbeing and how their value can be

translated into economic units. The following section introduces briefly the principal ecosystem services supplied by tropical forests, while contextualizing their importance in Central America. Problems derived from the ecosystem services classification are discussed in this section as well. I then analyze the benefits provided by ecosystem services to human wellbeing and the types of values associated with these benefits. I examine the main challenges in economic valuation, its implications in the long-term and the existing trade-offs in the monetization of ecosystem services. An overview of the main methodological approaches for economic valuation is provided in the penultimate section, where limitations and drawbacks are also examined. Finally, some concluding remarks are presented.

Ecosystem services and problems in their categorization

Ecosystem services relevant for tropical forests

In this section I provide a brief overview of the types of ecosystem services which are relevant for tropical forests, following in the first instance for simplicity the classification used by the Millennium Ecosystem Approach (MEA), which refers to the well-known categories of cultural, provisioning, regulating an supporting services.

Cultural services refer to a broad category of non-material benefits which include recreation and ecotourism, aesthetic, cultural diversity, spiritual and educational values, sense of place and identity, and inspiration. Recreation and ecotourism are of great importance in Central America given the profile of tourists approaching countries such as Costa Rica or Panama. These are well known destinations for ecotourism and rural tourism where visitors are attracted to large getaway houses and farms close to touristic centers, or for enjoying adventure sports, such as rafting and canopy. Other countries such as Honduras, Guatemala, El Salvador and Nicaragua also offer these types of tourism although their political instability has led to a lower expansion of this sector. Recreation in forested areas refers to activities that visitors do in their leisure time, such as walking, hiking, swimming and kayaking, camping, picnicking and enjoying scenic beauty, among others. Ecotourism is slightly different from recreation as it implies a sustainable use of natural resources, entailing sustainable benefits for local communities, support for conservation effort and low-impact activities. As defined by the International Ecotourism Society (TIES), ecotourism refers to a "responsible travel to natural areas that conserves the environment and improves the wellbeing of local people". Similarly, the World Conservation Union (IUCN) defines it as an "environmentally responsible travel to natural areas, in order to enjoy and appreciate nature (and accompanying cultural features, both past and present) that promote conservation, have a low visitor impact and provide for beneficially active socio-economic involvement of local people." Aesthetic services are linked to the benefits people obtain from the landscape, such as scenic beauty and can be seen as a part of recreational activities and ecotourism.

Apart from tourism, tropical forests hold high existence or passive use values related to the high biodiversity levels they sustain and their global importance as world's lungs. These benefits are not linked to any existing market in which they can be measured. They can be referred to as intangible values and for this reason are difficult to value. People obtain benefits from the existence of these forests even if they will never visit these areas (Adams *et al.* 2008). Some of these benefits, such as spirituality, education or inspiration fall in the category of cultural services, as stated in the MEA report (MEA 2005). However, the MEA is not clear as to specify whether existence values related to conservation purposes are or should be included in the cultural services category.

The category of *provisioning services* includes a wide variety of goods essential for human welfare such as food, fiber, fuel, biochemicals, natural medicines, pharmaceuticals and water supply. They are often categorized in terms of wood forest products (WFPs)[1] and non-wood forest products (NWFPs).[2] In tropical forests, NWFPs are often common property resources; they include a variety of fruits, nuts, seeds, oils, spices, resins, gums, medicinal plants and many more products specific to the particular areas from which they originate (Hassan and Ngwenya 2006). Bioprospecting is also considered as a provisioning service and is related to forest genetic resources and natural products for pharmaceutical or other industrial uses. The biodiversity maintained in tropical forests has indeed high potential for commercial use and contributes to private benefits linked to industries such as pharmaceuticals, cosmetics or biochemicals. This service has often been exploited in the form of Payments for Ecosystem Services, where companies sustain local communities to preserve forests in exchange for the right to operate in the forests looking for new products. Central American forests hold many examples of these arrangements and benefits vary depending on the industry and the country involved, but bioprospecting activities and revenues are expected to increase over the next decades (MEA 2005).

Finally, some of the provisioning services are of particular importance as they provide inhabitants with adaptive capability in the face of climate change. This is, for example, the case of forest as water providers. In the face to the expected decrease of precipitation in certain areas due to climate change, forest capacity for retaining and supplying water can increase the adaptive capacity of watershed communities. Freshwater provisioning (or water supply) is linked to processes related to filtering, retention and storage of water in streams, lakes and aquifers, which are accomplished through the soil and the vegetation cover. The storage capacity is very much dependent on the topography and sub-surface characteristics of the ecosystem. The water supply function focuses primarily on the storage capacity of forests rather than the flow of water through the system. Ecosystem services associated with water supply relate to the consumptive use of water by households, agriculture and industry (De Groot *et al.* 2002).

Another important category is that of *regulating services* which refer to the regulation of climate, water and biological plagues, as they act as a sink to greenhouse gases, they contribute to water quality and control for pests that could derive into diseases. Moreover, tropical forests have the capability to resist and

regulate natural hazards, such as tsunamis (in coastal areas), hurricanes and floods. With respect to forest fires, however, tropical forests have a very low regulating capability and in fact they are threatened by this risk.

Tropical forests are also very important sinks for carbon, mostly contained in the vegetation, while soils have much lower levels of carbon, because of the fast decomposition of dead biomass due to the climate conditions (Gibbs *et al.* 2007; Gorte 2009). The role of carbon sequestration and carbon storage is well known and has been extensively studied in the literature. The first is the process through which carbon dioxide is captured from the atmosphere and converted into biomass, above and below ground, while the second is the amount of carbon which is actually stocked in the biomass over the life cycle of the forest. In this process, deforestation accounts for large quantities of carbon emissions in the atmosphere. Although forests release carbon in the atmosphere at specific timing, they are net carbon sinks over their life cycle (Gorte 2009).

Tropical forests provide also essential services related to water, through the maintenance and regulation of the hydrological cycle. Certainly, the ability of healthy watersheds to moderate water flows and purify drinking water supplies is considered to be one of their most tangible and valuable services for humans (Postle *et al.* 2005). Ecosystem services derived from the water regulation function are, for example, the maintenance of natural irrigation and drainage, buffering of extremes in discharge of rivers (thus flood protection), regulation of channel flow, provision of a medium for transportation, groundwater recharge, water purification and erosion control. These services are obviously related with the provision of safe drinking water for the population and water supply for irrigation and hydropower. Countries in Central America are highly vulnerable as regards the supply of these services, due to the seasonality of precipitations and extreme events, as well as the presence of farmland located downstream of a river which is affected by the upstream hydrological process (Bonell and Bruijnzeel 2004).

Finally, *supporting services* act as a support for the other services to be maintained. Tropical forests have high rates of net primary production, which is directly related to the wood and non- wood forest provisioning services. Nutrient recycling is also essential for the functioning of the forest ecosystem, such as soil formation and others. Furthermore, any natural ecosystem (e.g. forests, water bodies, wetlands) provide living space and breeding/nursery areas for wild plants and animal species. Since these species are interrelated with the provision of environmental services, the maintenance of healthy (water) habitats is a necessary pre-condition for the provision of all ecosystems services, directly or indirectly. All these services are strictly linked to the ecosystem biological functioning and cannot therefore be directly associated to the production of welfare, so that they are usually not included in the economic valuation to avoid double counting.

Beyond the MEA classification

The MEA approach (MEA 2005) and the emphasis given on ecosystem services has received a wide recognition by the scientific and policy communities, mainly

because it relates the physical and biological aspects of ecosystem functioning (and biodiversity) with the service provision and human welfare. In this sense it provides useful inputs for translating the impacts on biophysical assets derived from competing use of resources into socio-economic impacts. Certainly, the focus on ecosystem services has provided new approaches for research as well as for communication and educational purposes. There are still, however, many uncertainties surrounding the ecosystem services assessments. Limitations stem from different perspectives. The dynamics of biological systems and their interactions with social organizations are still unknown, and basic information is lacking. From a biological and ecological perspective, rigorous assessments require to address issues related to non-linearity in the provision of ecosystem services, thresholds effects of non-linear changes, spatial and temporal variability, relations between dynamic and adaptive systems, irreversible changes and the existence of complex feedbacks going beyond causal interactions (Koch *et al.* 2009; Paavola and Hubacek 2013; Serna-Chavez *et al.* 2014; Bagstad *et al.* 2014).

On the other side, the classification proposed by MEA has raised many criticisms, while there is a confusion deriving from the terminology used and related to the notions of "functions", "services" and "benefits," which are often employed with diverse connotations. This is particularly relevant for the economic valuation of ecosystem services, as the MEA classification can lead to double counting. The problem may arise when a service is valued at two different stages of the same process (Bateman *et al.* 2011; Boyd and Banzhaf 2007; Fisher *et al.* 2009). In this context, many studies have questioned what should be the best classification of ecosystem services when economic valuation is the goal of the analysis (Bateman *et al.* 2011; Boyd and Banzhaf 2007; Wallace 2007; Fisher and Turner 2008). As a result, differences exist, for practical reasons, on the classification and measurement of ecosystem services among studies.

A good example of the double counting issue can be seen for water-related services. In fact, forests provide water flow which is a regulating service, and also water supply for hydropower which is a provisioning service. The benefits from water supply for hydropower are directly dependent on the flow of water, so that if both services are assessed in monetary terms, the benefits obtained from that ecosystem would be overstated. To address this issue, the approach proposed by Ojea *et al.* (2012) focuses on the use of an output-based classification as an alternative to MEA. The main problem in the MEA classification and others that have been proposed (De Groot *et al.* 2002; Alyward 2002) is that they mix up different concepts of ecosystem services referring alternatively to structural factors, processes and outcomes or benefits.[3] Yet, for the economic valuation, it is rather the outcome of the ecological process that should be assessed and not the process. The outcome of the ecological process is the product or service that the ecosystem provides to humans.

In this context, Ojea *et al.* (2012) propose an adaptation of the classification of water-related services used by Brauman *et al.* (2007), based on the outcome, and where water-associated supporting services[4] are excluded: (i) improvement of extractive water supply, (ii) improvement of in-stream water supply, (iii) water

damage mitigation, and (iv) provision of water-related cultural services. Under this classification, water supply is a provisioning service describing ecosystems modification of water used for extractive and in-situ purposes, which include municipal, agricultural, commercial and industrial use. In-stream water supply includes hydropower generation, water recreation and transportation, and freshwater fish production. Water damage mitigation is a regulating service; it includes mitigation of flood damage, sedimentation of water bodies, saltwater intrusion into groundwater and dry-land salinization. Cultural hydrologic services include spiritual uses, aesthetic appreciation and tourism.

This approach avoids the problems related to double counting when the same service is valued in different stages of the ecological process. But it also allows, in an operative manner, to integrate the value of one same service when it is valued at two different stages of the same process providing human welfare. For example, under this classification, water flow (which is essentially a regulating service) and water supply for hydropower (which, according to the MEA classification, may be considered as a provisioning service) will be included under the same category: improvement of in-stream water supply. As Brauman *et al.* (2007) point out, ecosystem services can therefore be assessed at different stages of production, by quantifying the magnitude of attributes or intermediate services levels, or by assessing the amount of final service benefits.

Another approach which has been put forward to overcome the problem related to the ecosystem services-based classification (and related issues of double counting) is to emphasize the beneficiaries and the spatial dynamics of ecosystem services (Villa *et al.* 2014; Serna-Chavez *et al.* 2014). Villa *et al.* (2014) use the ARIES (Artificial Intelligence for Ecosystem Services) methodology to spatially map the beneficiaries (or users) and consequently the ecosystem service flows benefitting specific and concrete geographical areas. Serna-Chavez *et al.* (2014) propose a framework to analyze the spatial dynamics between the areas of ecosystem services provision and the beneficiary areas where the services are consumed. These approaches allow an improved spatial and temporal characterization of the flows of ecosystem services, while computing the underlying processes, thus solving the issue of double counting and related quantification of processes and outcomes or benefits.

Economic valuation and types of benefits to humans

Ecosystem service valuation: context and challenges

Ecosystem services have been described and expressed in terms of the benefits that people derive from the ecosystems (Daily 1997). According to the mainstream economics, these benefits are contributing to human wellbeing and health; they are linked to human welfare and can therefore be translated into economic values, which are associated with individual preferences. When we speak of economic values we mean values which are expressed in monetary units (based on the people's willingness to pay) even when a market does not exist. The value of

ecosystem services reflects basically how much money the society is willing to allocate to protect them, which becomes crucial in a world of scarce and limited resources, and especially in a context of global change with multiple stressors, system interactions and trade-offs. Economic values are critical to carry out cost-benefit analysis to inform decision-making process about the economic and financial appropriateness of planned interventions. They allow also comparing different types of services with a common unit of measurement, though there are many criticisms that should be highlighted.

This emphasis on monetary values has developed within a utilitarian and anthropocentric approach,[5] in a broader context aiming at reconciling economic growth with environmental and biodiversity conservation (TEEB 2010). It is also reflected in recent movements, such as the Convention on Biological Diversity (CBD) and the MEA. According to some authors, economic valuation and the reference to monetary values have, therefore, to be seen within the general political and economic context designated as "neoliberalism" (Gómez-Baggethun and Ruiz Pérez 2011). In addition, failure of traditional conservation approaches to protect ecosystems and biodiversity have urged solutions such as the use of economic valuation as possible ways to overcome the problem.

There are, however, a number of issues and shortcomings which need to be highlighted in the context of ecosystem services valuation. First, there is the question of commodification of nature, which refers to the pricing of natural resources and processes, and their consequent inclusion in the market as tangible objects for commercialization and privatization. The concept poses ethical and equity concerns as it promotes the establishment of property rights (within a privatization perspective) for things which are public goods and for which people do not have necessarily a priori monetary values. The establishment of property rights is encouraged by the fact of making these services exchangeable through the market, though originally not intended for sale. An example is given by the Protocol's Clean Development Mechanism (CDM) where carbon credits are traded in emissions trading schemes. In this sense, the process of commodification promotes social inequalities as the access to ecosystem services would be subject to a pricing system and to the individual predisposition to pay for it. Another issue is linked to the complexity of the ecosystem services which, under the commodification process, would become discrete making use of comparable units of measurement. Though commodification has to be detached from the concept of economic valuation, the latter operates as a prerequisite for commodification while endorsing its practice (Gómez-Baggethun and Ruiz Pérez 2011).

A second set of problems is related to the presence of intangible and intrinsic benefits in the ecosystem, which are hardly or impossible to quantify in terms of willingness to pay. Intangible benefits are related to services such as cultural identity, spiritual values and similar, while intrinsic benefits are not quantifiable as they are totally independent from the needs of human beings. In this sense they depart from a utilitarian perspective.

Third, monetary values depend strongly on the socio-economic context where the valuation process is carried out, in terms of individual perceptions (subjective

values) and influence of institutional, political and cultural factors.

A fourth issue is related to the fact that economic valuation cannot be used when approaching the ecological threshold, where the provision of ecosystem services falls off disproportionally with risks of irreversible changes (Wegner and Pascual 2011). It must be added that ecosystem services are usually not substitutable services or only to some limited extent, and that economic values are meaningful only when the service provision is stable and marginal changes are predictable.

For all these reasons, recent trends have moved towards more comprehensive approaches integrating multidimensional aspects in order to incorporate other types of values in the ecosystem assessment, such as social or collective values (TEEB 2010; O'Connor *et al.* 1998). In this context, examples of integrated approaches are given by deliberative valuation and multi-criteria analysis (Wegner and Pascual 2011). Deliberative valuation is built on the idea that preferences for the environment should not be based on individual pre-determined perceptions that are aggregated afterwards. Instead, these preferences should be built through a deliberative process with the participation of different groups of social actors interacting to analyze the problem, exchange the information and find an agreement. Multi-criteria analysis is a multidimensional approach used for decision-making, which compares different alternatives and preferences using specific indicators, where the economic value or price is just one of them. Social, environmental and economic factors are all considered in the analysis, allowing therefore for multiple value types.

Another more innovative approach is suggested by Villa *et al.* (2014) who propose the use of artificial intelligence and specifically a modelling platform combining complex models to map ecosystem services flows and their beneficiaries, building on dynamic spatial-temporal processes and probabilistic analysis (e.g. ARIES). This platform does not currently include a model for monetary values, though it could be adjusted to include a value-transfer function.

In conclusion, in spite of the many limitations and non-inclusive aspects of economic valuation, its contribution as a tool of communication and awareness is still recognized, as it provides a feedback on the negative effects associated with the over-exploitation of natural resources. Furthermore, it represents a useful tool for decision-making when planning interventions for biodiversity protection and conservation, as it allows to measure the impacts and benefits and to make, therefore, final choices about budget allocation among different objectives. The monetized benefits of biodiversity can in fact be compared with the costs of different protective measures and help in setting taxes for examples. Monetary values are also useful for making appropriate land use decisions and to set limitations in trade in endangered species. Finally, as highlighted also by The Economics of Ecosystems and Biodiversity study (TEEB 2009), most of the services provided by the ecosystems are not captured by conventional macro-economic indicators, such as the gross domestic product, as they are not traded in regular markets. As such, they cannot be taken into account in conventional accounting systems such as the SNA (Standard National Accounts). The next section focuses on the types of

economic values which have been proposed in the literature to measure the benefits provided by the ecosystem.

Types of economic values

The well-known concept of total economic value (TEV) is an attempt to consider the multiple facets of the benefits provided by the ecosystem. The TEV refers to the aggregated economic value of benefit flows generated by the ecosystem and comprises different categories of value types, usually referred as use and non-use values (Krutilla 1967). It reflects a discounted value of the current and future flows of benefits. Biodiversity and ecosystems contribute basically to the different elements of the total economic value, by increasing the human welfare directly or indirectly.

Use values can include both *direct* and *indirect use* of a resource. The first represents the actual or planned use of an ecosystem service, in the form of consumptive use (e.g. food, timber, plants) or non-consumptive use (recreation, landscape amenity and aesthetic). The second (*indirect use*) denotes the benefits derived from ecosystem services that are supported by a resource such as water regulation, flood control or pollination (De Groot *et al.* 2002). *Option values* can be also part of use values where referring to the value that people place on having the option to use a resource in the future, even if they are not current users. On the other side, *non-use values* are those benefits that are not associated to a specific use, such as the existence value of a resource or its spiritual value. This broad category includes quasi-option, bequest and existence values. The *quasi-option value* refers to the benefits obtained from future information made available through the preservation of the natural resource. *Bequest values* include the benefits provided by the resource for future generations, while *existence values* is related to the satisfaction that people receive from knowing that a resource exists, though in the absence of direct enjoyment. A number of methodological approaches have been developed in the literature in order to estimate these values, as it is presented and discussed in the next section.

Provisioning services are always related to direct use or option values as they provide products frequently with commercial use or for human consumption. Option values can arise as well if we consider the current value of a forest stock, and essentially in the case of bio-prospecting where private companies can be interested in conserving the biodiversity of a forest for the potential benefits they can derive in case a new product is obtained from plant species. Cultural and amenity services are clearly related to direct use and option value when assessing recreation, since actual and potential visitors can be surveyed. In the case of spiritual services, both existence and option values can be considered, besides the direct use, as some people may place a non-use value to the forest today, or keep the option of consuming it in the future. Finally, regulating services are related to both direct and indirect uses. We refer to direct use if, for instance, a hydroelectric power company is benefiting from the forest hydrological regulation, while the indirect use would refer to the benefits that people receive from water retention.

Existence values may arise also for regulating services, for example, if people value the benefits of a watershed in terms of water quality with no relation to its consumption, because they consider that the ecosystem should be in good health for its mere existence. Though the TEV is the most accurate and comprehensive measure to consider for the monetary assessment of ecosystem (services as it accounts for all benefits provided by the service itself), in practice this measure has not been fully integrated in the economic valuation framework. As a consequence, the different components of the TEV are not always expressed in monetary terms, although in principle they could be, which results in an underestimation of the true value of the service.

Valuation methods and their applicability for ecosystem services

The economic valuation is anchored to the human welfare. A change in the quantity or characteristics of the ecosystem service has an impact on human welfare and can, therefore, be valued in monetary terms. In a world of limited resources, the economic value of ecosystem services reflects how much the society is prepared to pay in order to protect and conserve the ecosystem (TEEB 2010). In this context, it is assumed that a loss of service will result in a loss of welfare, which can be translated into a cost for the society. This viewpoint is crucial for appropriate decision-making and a reasonable allocation of the resources.

The objective of the economic valuation is to estimate incremental or marginal effects, given by variations in stocks or assets. For comparability reasons among ecosystem services, a common unit of measurement is usually employed, such as the value per hectare per year which has been extensively used in the literature. All the values mentioned in the previous section can be measured in monetary terms using different techniques. Broadly speaking, there are two main approaches potentially applicable to the valuation of forest ecosystem services: market- and non-market based methods (Table 2.1). The first make use of market prices and data, and they are applicable in the following cases: (i) when the ecosystem service is traded in the market (*market price method*); (ii) when the service contributes to the production of a marketed good or service (*production function*), (iii) when the service can be replaced by alternative goods and services having market prices or costs, or when an action could be taken to avoid damages (*cost-based methods*). An exchange of goods and services through the market is the reference point in this group of methods.

The *market price method* calculates the marginal value of the service using market prices and databases. This is feasible for those ecosystem services which have a price in existing markets, such as wood forest products (e.g. timber), pharmaceutical contracts and financial revenues from outdoor recreational activities. In a competitive market without distortions (e.g. taxes or subsides) the price is determined by the relative demand for and supply of the good or service in question, and reflects its marginal value. The *production function* is used to estimate the value of a non-marketed service by assessing its contribution as an input into the production process of a commercially marketed good (Ricketts *et al.*

Table 2.1 Valuation methods and approaches for monetary assessment

Approach	*Method*	
Market valuation	Market price methods	Market prices
	Production-based methods	Production function
	Cost-based methods	Replacement cost Damage cost avoided Averting behaviour Net factor income
Non-market valuation	Revealed preferences	Travel cost Hedonic pricing
	Stated preferences	Contingent valuation Choice experiment

2004; Núñez *et al.* 2006; Barbier 2007). A production function describes the relationship between inputs and outputs in production, such as the production of hydropower generated from flowing water (see Chapter 4 for an application). Given a change in the ecosystem input, by calculating the change in the value of production, it is possible to observe the value of that input using the coefficients estimated by the function. Finally, the *cost-based methods* include a variety of techniques focusing either on the cost of replacing the service with alternative man-made goods and services (*replacement cost*; Byström 2000; Su and Zhang 2007), or on the cost of actions taken to avoid damages (*damage cost avoided*; Hein and Gatzweiler 2006; Barbier 2007), or on the expenditures supported to avert damages (*averting behaviour*; Cropper and Freeman 1991), or on the inputs and revenues of a marketed good to which the service contributes (*net factor income*; van Beukering *et al.* 2007).

The second approach for valuation refers to non-market methods which employ hypothetical markets and willingness to pay measure to derive a monetary value of the ecosystem service, through *revealed* and *stated preferences* (Braden and Kolstad 1991; Freeman 1993; Carson *et al.* 1996; Dickie 2003). *Revealed preferences* are based on the observation of actual consumer or producer behavior, and they identify the ways in which a non-marketed service influences actual markets for some other goods and services. Preferences and values are "revealed" through consumers' choices in complementary or surrogate markets. This could be done by applying the *travel cost approach* (visitors incurring expenses to visit natural areas; Christie *et al.* 2007), or the *hedonic price* (looking at the housing market to derive environmental quality values; Humavindu and Stage 2003). On the other hand, *stated preferences* are used for those services which are not traded in the market and thus have no market price. They are based on surveys and collect data to estimate the individual willingness to pay for a change in the level of provision of the service. These methods are the only ones capable of deriving economic estimates for high existence values or cultural values not related to direct use

(Bateman *et al.* 2002). There are several methods in this category including the *contingent valuation method* (where respondents are asked for their willingness to pay for specific changes in environmental quality; Arrow *et al.* 1993; Nunes and van den Bergh 2001), and the *choice experiment method* (where respondents are asked to rank the service attributes or to choose among alternative scenarios; Hanemann 1994; Hensher *et al.* 2005).

As the implementation of these approaches is sometimes unviable due to the amount of data needed and the consequent financial resources requested, researchers often use an alternative approach, known as *benefit transfer*. With this method, data of existing studies, estimated through market and non-market methods, are adjusted and transferred from the original locations to new locations and contexts. Basically, values are transferred from the "case study sites," estimated in the original studies, to "policy sites" where no reference study exist. The benefit transfer is used to scale up values from local sites to larger geographical scales (Wilson and Hoehn 2006; Chiabai *et al.* 2011).

The benefit transfer approach has been highlighted as a "broad-scale assessment" (Nelson *et al.* 2009) where it is assumed an equal value for each hectare of the ecosystem under valuation. This assumption is, however, quite simplistic as it does not take into account the quality of the ecosystem, its total extension, and the types of ecosystem services supplied. It is, in fact, incorrect to attribute the same economic value to each hectare of the land, as the flow of services is not directly proportional to the size of the land, even in normal situations distant from the ecological threshold. In addition, there is a critical size of the habitat for the provision of ecosystem services and reductions below that size would threaten the whole habitat. Nevertheless, to avoid complications in the estimation process, in the literature the economic valuation is always performed using the value per hectare as unit of measurement.

In the valuation exercise the choice of the valuation method becomes crucial and will depend on the type of service under valuation and the types of values that need to be assessed. For provisioning services, market methods are often applied, but also hedonic pricing techniques can be used. A variety of methods can be applied for the valuation of regulating services, but there is a predominance of revealed preferences methods such as the avoided and replacement costs. Cultural and amenity services require the use of stated preferences when non-use values are to be valued. Recreational and aesthetic values can be measured by revealed preference techniques, while non-use values (spiritual, identity and similar) necessarily require the use of stated preference techniques.

Given the issues and uncertainties in the monetary valuation and the intrinsic limitations in each technique discussed above, the need for non-economic tools has arisen (Ormsby and Kaplin 2005). These can be based on the administration of qualitative surveys where people's opinions, attitudes, experiences and perceptions about biodiversity and ecosystem services are gathered and recorded (Christie *et al.* 2012). Other techniques focus on deliberative and participatory approaches, such as *action research* and *Delphi surveys*. *Action research* is used in the planning, implementation and monitoring of projects for the community, and aims

at actively engaging the target population in the research process (Barton *et al.* 1997). The *Delphi method* is based on structured surveys and iterative processes, where a panel of experts is interviewed through a questionnaire subject to more rounds with the possibility of revisions in each of them (Taylor and Ryder 2003). All these technique, though offering useful information, do not provide economic values, but only descriptive analysis of people's perceptions and preferences for ecosystem services and biodiversity.

Conclusions

This chapter offers an overview of the importance of ecosystem services provided by tropical forests, how these are linked to human wellbeing, to what extent economic valuation can be a valid instrument and which are the challenges to face.

The focus on ecosystem services has been widely acknowledged in the literature as an important instrument to drive research and policy decisions (Carpenter *et al.* 2009). However, robust approaches and meticulous analysis are needed to provide valuable information to policy-making, though lack of data is a key question. Issues that should be considered in future research include, among others, the necessity of addressing non-linear and irreversible changes, which can lead to long-term impacts on ecosystem services flows, interactions and feedback mechanisms between ecological and social thresholds, trade-offs between ecosystem services (the overuse of some services might decrease the availability of others), spatial and temporal displacement of benefits and costs, and multi-scale interactions. Another key factor is the economic valuation and how this should be conducted in order to provide useful information for decision-making. Box 2.1 summarizes its main challenges in a policy context.

Box 2.1 Challenges for the application of economic valuation in a policy context

How and to what extent can economic valuation provide useful and meaningful information for policy-makers?

1. *Involvement of local policy-makers.* In the first instance, it is necessary that the research produced is able to link with local policy-making and this is possible only if local researchers are involved in the study and if participatory methods are used (Christie *et al.* 2012). In order to provide effective and durable results, participation has to be fostered within local communities so that information learning is promoted and values can be truly elicited in more inclusive patterns. This paves the way for higher and more effective involvement of social groups in research and decision-making. Once established within a similar context, it is possible to look for new processes for incorporating participation into economic valuation (e.g. deliberative

processes and valuation or multi-criteria analysis).

2. *Identification of relevant stakeholders and beneficiary groups*. A further point, which is closely linked to the previous one, concerns the need to identify all relevant stakeholders for each ecosystem service and involving them in the process of economic valuation itself (TEEB 2010). This process entails different steps embracing the identification of the services affected, their valuation and recognition of feedback mechanisms, ecological thresholds and trade-offs, as well as social conflicts and equity considerations. Another crucial aspect for improving the accuracy of economic assessment is the necessity of mapping the groups who benefit from the provision of ecosystem services as well as the spatial dynamics of the flows. The formation of economic values should also reflect the appropriate spatial scale depending on the type of ecosystem service analyzed. However, in some cases, both local and broader spatial scales (e.g. national/international) should be contemplated as the values emerging from them can be substantially different. For example values emerging at global level may offset those arising at a more local scale. This can be fostered by involving stakeholders from different geographical dimensions in an iterative participatory process.
3. *Identification of ecological thresholds in the service provision*. It becomes important to recognize when an ecological system is near to its ecological threshold as at this point non-linear effects are expected to come into existence and economic valuation becomes nonsense.
4. *Influence of future scenarios of changes (e.g. climatic, socio-economic)*. The influence of future socio-economic drivers and scenarios is another factor to take into account, as important changes can be expected in terms of demographic projections, urbanization, income and energy demand, as highlighted in Chapter 7. In other terms, how economic values will change under these future scenarios is a key issue because the socio-economic impacts might even dominate the bio-physical impacts.
5. *Uncertainty analysis*. Finally, uncertainty analysis should be incorporated in the economic valuation (for individual preferences and methodological approaches) as well as in the bio-physical assessments (as regards the provision of ecosystem services flows).

Considering these challenges, economic valuation can be used as a tool to influence decision-making when a choice about the use of resources has to be made. It becomes, indeed, a key factor in setting up conservation strategies and can influence policies for adaptation, despite the many limitations and the fact that many aspects cannot be covered. In this context, cost–benefit analysis can provide useful information for trade-offs analysis and to identify win-win options among multiple alternatives. It can be useful also to establish compensation mechanisms

to remediate damages for example or to establish economic instruments such as payment for ecosystem services (PES). As a further point, economic valuation is beneficial to support informative processes within decision-making or to raise awareness in the public opinion about conservation purposes. In all these contexts, however, it is necessary that economic valuation is strongly anchored to biophysical modelling and outputs. This is presented in two different settings in the present book, where the economic assessment is framed on biophysical models, with an attempt to address some of the key challenges in the economic valuation. In the first setting (Chapters 4 and 5), biophysical and economic models are combined within a common framework, where the biophysical impacts of climate change on ecosystem services are translated into monetary terms, using spatial analysis, output-based assessments, production function, meta-analysis and benefit-transfer. The second setting (Chapter 6) incorporates land-use change modelling and spatial data into cost–benefit analysis to evaluate alternative land-use options affecting differently forest ecosystem services.

Finally, alternative approaches to economic valuation have been also suggested in the literature to overcome some of its limitations and incorporate multidimensional aspects and values. These include for example deliberative valuation and multi-criteria analysis which embrace social and collective values (Wegner and Pascual 2011), or more innovative approaches such as the use of complex modelling platforms based on artificial intelligence (Villa *et al.* 2014) and quantitative frameworks (Serna-Chavez *et al.* 2014) to assess spatial flows of ecosystem services and map beneficiaries. Other approaches have adopted spatial modelling tools integrating production functions and economic valuation to analyze trade-offs between ecosystem services and overcome some of the limitations recognized in the "broad scale assessment of multiple services" (integrated valuation of ecosystem services and tradeoffs; Nelson *et al.* 2009; Goldstein *et al.* 2012). An interesting framework is the one proposed by Villamagna and Giesecke (2014) who suggest to use human wellbeing (HWB) measures reflecting multidimensional aspects of wellbeing, and to analyze how these measures are affected by changes in the provision of ecosystem services.

Future research should explore the conditions for integrating these approaches within an economic valuation framework with the aim of addressing key issues such as spatial and temporal scales, trade-offs analysis, use of appropriate units of ecosystem services, non-linearity between loss of forest and corresponding loss of services flows. These aspects translate into the need of understanding how economic values should change when taking into account the quality of the ecosystem (so as to avoid using systematically an equal value for each hectare of forest lost), or when approaching critical ecological thresholds.

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Notes

1 Wood forest products can be grouped into different categories corresponding to industrial sectors in the market (FAO/ForestSTAT), namely industrial roundwood, wood pulp, recovered paper, sawnwood, wood-based panels, paper and paper board, and wood fuel.

2 Non-wood forest products comprise items such as wild food that are considered agricultural products as well as raw materials for crafts and construction.

3 Specific to water services, Lele (2009) points out that it is necessary to distinguish between the structure of the ecosystem, the ecosystem processes and impacts (or outcomes). In the context of tropical forest, the author says, structural change takes many different forms (e.g. from intact natural forest to heavily hacked thickets, from natural forest to pasture or timber plantations, etc.). Each of these changes can influence several watershed processes (e.g. erosion rates, sediment load, water chemistry, peak flow levels, total flow, base flow or ground water recharge) in different ways. These changes can result in different kinds of human impacts (e.g. increased costs of water purification, increased fertilization of floodplain lands, changes in agriculture, reduced drinkable water etc.). These impacts can affect different stakeholders (farmers, drinking water users, livestock owners, floodplain residents, hydropower companies) and can be positive or negative. According to this distinction, the process would not be the "service," but the human impact would.

4 The water-related supporting services of terrestrial ecosystems are wide-ranging and include the provision of water for plant growth and habitat for aquatic organisms such as estuaries.

5 The utilitarian approach is grounded on the Cartesian and Newtonian system in which the universe is compared to a mechanical clock ("clowork universe theory"; TEEB 2010).

Part II

Climate, water and land-use changes in Central American tropical forests

3 Impacts of climate change on ecosystem hydrological services of Central America

Water availability

Pablo Imbach, Bruno Locatelli, Juan Carlos Zamora, Emily Fung, Lluis Calderer, Luis Molina and Philippe Ciais

Introduction

Climate change and land use change are predicted to be the two main drivers of global biodiversity alterations (Sala *et al.* 2000). The effects of climate change on biodiversity are manifested at different scales, from genes (i.e. generating mutations and simplifying gene pools) to species (i.e. extinctions and range modifications) and ecosystems (i.e. community composition and distribution changes). These effects influence ecosystem functions and processes, for instance changes in the abundance and composition of plant species may influence water cycle, nutrient and carbon dynamics, trophic interactions and disturbance regimes (Chapin III *et al.* 1997; Diaz *et al.* 2004).

Ecosystem functions are linked to the provision of ecosystem goods and services when human values are considered (de Groot *et al.* 2002; MEA 2005). Many authors have highlighted the value of these ecosystem services for sustainable development, globally (Costanza *et al.* 1997) or locally (Lutz *et al.* 2000; Woodward and Wui 2001; Pattanayak 2004). Terrestrial ecosystems provide an array of watershed services, for instance the regulation of hydrological flows, benefiting agriculture, drinking water users, energy production, or transportation (Costanza *et al.* 1997; MEA 2005).

Watershed services are of outmost importance in many developing countries where water is a crucial development issue, such as Central America (Locatelli *et al.* 2010). According to UNDP (2006), around 20 per cent of global population lives without access to potable water. Nicaragua being the second poorest country in Latin America provides a good example, where only 49 per cent of the rural population has access to this resource and high water-related mortality is observed (CONAPAS 2006). Potential conflicts in the future could be expected due to freshwater use and access, given pressures from population growth and water use demand (i.e. for irrigation and industry; UNFPA 2006).

The degradation of ecosystems and the associated loss in the provision of ecosystem services are major threats for human well-being (MEA 2005). Knowing

about the impacts of climate change on the ecosystems hydrological functions is necessary to inform policy makers about the risks induced by climate change and support their decisions about adaptation to future changes (Scholze *et al*. 2006).

In this chapter we aim at assessing changes in ecosystem hydrological services under future climate scenarios and impacts on per-capita water availability for main watersheds of Mesoamerica. We used several climate change scenarios and a biogeography model to project a range of potential changes in the provision of ecosystem services while accounting for uncertainty from climate models and future emission scenarios. These results could be used as the basis to assess the vulnerability to climate change, at the national and regional scale, of important economic sectors relying on hydrological services (i.e. agriculture, power generation, drinking water).

Previous studies in Central America

Climate change scenarios

Changes in atmospheric concentration of greenhouse gases since the beginning of the industrial revolution have modified the natural dynamics of the global climate. The range of potential future storylines suggests that the trend will continue and in most scenarios further increase. The development of future climate scenarios relies, generally, on three components:

- the historical and future emissions of greenhouse gases and aerosols into the atmosphere (emission storylines);
- the simulation of climate models accounting for global warming as a result of changes in radiative forcing (the energy balance between the radiant energy received by the Earth from the Sun and radiated back to space) associated to concentration trends of these greenhouse gases and aerosols (radiative effects) and to land use changes (biophysical effects); and
- the regional climate change resulting from this forcing.

Emission storylines and the future increase in greenhouse gases and changes in aerosols depend on the evolution of several socio-economic parameters (i.e. population, land use or technology) as well as climate policy efforts to meet radiative forcing targets. The IPCC (Intergovernmental Panel on Climate Change) has developed a range of storylines or "emission scenarios" up to the year 2100 (IPCC 2000), which were used to force climate models (referred as SRES for Special Report on Emission Scenarios). These storylines framed emissions scenarios used to simulate future climate in IPCC Fourth Assessment Report (IPCC 2007) and set the basis for the representative concentration pathways (RCP) (and resulting radiative forcing) used to simulate climate scenarios with the latest generation of climate models for its Fifth Assessment Report (IPCC 2013). The current generation of global climate models (GCMs; or general circulation models) integrate atmosphere, ocean and terrestrial processes and simulate climate trends at large scales. The recent 5th IPCC Assessment is based on GCM

simulations from the fifth phase of the Coupled Model Intercomparison Project using RCP of greenhouse gases (Rogelj *et al.* 2012) that include effects of climate policy scenarios for the twenty-first century on emissions while the previous assessment used emission scenarios that did not account for future effects of mitigation and adaptation policies. Differences in climate scenarios between the 4th and 5th IPCC reports indicate, for example, a median global temperature increase of 2.4°C for 2090–2099 under RCP 4.5 (used in this study) and SRES B1 scenarios (3.4°C and 3.9°C for A1B and A2, respectively). RCP 4.5 median temperatures also rise faster until 2050 compared to SRES B1 (Rogelj *et al.* 2012). Given uncertainties in future emissions of greenhouse gases as well as in climate modeling efforts, it is important to assess future climate uncertainties when evaluating potential impacts of climate change. Furthermore, assessments of climate trends, for example based on weather stations or satellite data, allows for comparing these in future simulations with historical observations.

Weather station data and satellite observation allow comparing recent observed trends with the projected changes in future climate. Although, human caused changes in climate are difficult to detect and attribute from natural climate variability, trends in key climate variables have been already observed in Central America. Aguilar *et al.* (2005), for example, found an increase in temperature and precipitation intensity based on weather station data in the last half of the 20th century across the region. Malhi and Wright (2004) found a similar trend in temperature and an increase in annual mean precipitation in some parts of northern Central America. The authors also highlight the difficulty in discerning trends in precipitation across tropical areas due to high inter-annual variability.

A consistent drying signal in Central America's future climate is found across the existing range of climate scenarios, showing a general decrease in precipitation and an increase in temperature (Neelin *et al.* 2006). At the seasonal scale this trend could further reduce precipitation and length of the mid-summer drought (a period of reduced precipitation within the rainy season; Magaña *et al.* 1999), particularly in the central part of Central America (Rauscher *et al.* 2008). The increased temperature signal under future scenarios is consistent across global climate models while the change in precipitation shows higher disagreement with positive and negative anomalies depending on the model (Imbach *et al.* 2012). The expected magnitude of change in climate makes the region a climate change hotspot among tropical areas (Giorgi 2006) where mean temperature will move outside its historical variability envelope relatively sooner than other land areas (Hawkins and Sutton 2012; Mora *et al.* 2013).

The spatial configuration of the region, a topographically complex narrow strip of land, requires downscaling of global climate models into high resolution scenarios. But only a small set of downscaled scenarios are available so far. Karmalkar *et al.* (2011) developed downscaled scenarios (0.22° or ~25 km) using the PRECIS (Providing Regional Climates for Impacts Studies) model derived from HadRM3, the Hadley Centre Regional Climate Model (Jones *et al.* 2004). Nakaegawa *et al.* (2013a) used MRI (Meteorological Research Institute model of Japan) to develop scenarios at 60 and 180 km, with an ensemble of realizations, as

well as one simulation at 20 km. Both authors found improvements in simulating high resolution precipitation (compared to global simulations) with a wet/dry bias in the dry/wet seasons with PRECIS (MRI results focused on upper atmosphere dynamics that are out of our scope). Under high emission scenarios (A2; 2070–2100) Karmalkar *et al.* (2011) found higher warming during the wet season (relative to the dry season). Precipitation showed larger reductions during the wet season, and during the dry season over areas influenced by orographic precipitation (Pacific watershed). Nakaegawa *et al.* (2013b) found an increase in maximum 5-day precipitation and number of consecutive dry days with consistent changes over the Yucatán Peninsula and Guatemala (for consecutive dry days only).

Hydrological modeling

The Mapped Atmosphere Plant Soil System (MAPSS) model (a brief description is provided in the methods section) has been previously calibrated and water balance and leaf area index (LAI) outputs validated with historical observations for Mesoamerica (Imbach *et al.* 2010). The runoff output was validated against long-term average runoff from a set of catchments (n = 138) that covered the regional gradient of precipitation, elevation, catchment size and forest land cover. This model satisfactorily predicted annual runoff (with an under-estimation of 12 per cent) and the prediction of seasonal stream discharge was also well reproduced in 78 per cent of the catchments. The absolute runoff at the monthly time scale had a lower performance, with a satisfactory prediction only over 48 per cent of the catchments, probably due to model lack of capacity to simulate aquifers recharge and discharge processes across seasons. The LAI output was validated against two long-term average LAI maps from remote sensing sources. The model under-represents LAI values in the northern part of the region and over-represents LAI in the southern part (Costa Rica and Panama) that could lead to potential biases when modeling water balance. These differences are probably related to, besides model bias, high-cloud coverage that reduces the LAI algorithm performance from remote sensing sources. The authors recommended using MAPSS output values at the annual scale for further applications in Mesoamerica. Based on the same model setup, Imbach *et al.* (2012) analyzed uncertainties of discharge and vegetation distribution, under a range of future climate scenarios (average climate for 2070–2099) providing the starting point of the analysis presented here.

Hidalgo *et al.* (2013) used VIC (variable infiltration capacity), a macro-scale hydrological model, to assess impacts of climate change on Central American hydrology with special focus on drought prevalence. The model simulates hydrological variables from the surface (i.e. surface runoff, soil moisture, base flow) and energy balance near the surface (i.e. to derive evapotranspiration) using daily climate data. The model has a parameterization for soils, vegetation and snow distribution (not relevant for our study area), that was calibrated using and automatic procedure. Contrary to MAPSS the VIC model has a prescribed mosaic of vegetation, including disturbed land cover types (i.e. agriculture) but can assess transient changes (MAPSS estimates long-term average conditions).

Both VIC and MAPSS produced similar results, with water availability being *likely* reduced across most of the region (>61 per cent of Mesoamerica; Imbach *et al.* 2012) and drought frequency increases (Hidalgo *et al.* 2013). Imbach *et al.* (2012) also assessed associated changes in ecosystems and found *likely* reductions in leaf area index and increased evapotranspiration (the later in southern Mesoamerica) coincident with runoff reductions.

Water availability indicators at coarse scales

Several indicators have been developed to assess water availability at coarse scales. The water scarcity index (WSI) is simply defined as the fraction of total annual runoff available per capita (i.e. m^3 per capita) and helps distinguish climatic from human causes (i.e. poor infrastructure). The indicator was applied to multiple countries and used to categorize yearly water availability as: *no stress*, *stress*, *scarcity* and *absolute scarcity* (>1700, 1700–1000, 1000–500 and <500 m^3 per capita, respectively; Falkenmark 1989). It is commonly used at country level assessments and tends to under estimate water scarcity for smaller populations. WSI cannot capture differences of water requirements due to different lifestyles (linked to the intensity of use of the resource) or seasonal limitations on the resource (Brown and Matlock 2011).

Other indexes were developed to account for different human requirements (drinking, sanitation, bathing and food preparation needs) indicating a minimum water requirement of 50 liters per person per day (Gleick 1996). The *social resource water stress/scarcity index* evaluates the capacity of a society to adapt to changes in water availability, using indicators such as the Human Development Index as a weighing factor for adaptive capacity (Ohlsson 2000). The *ratio of annual water withdrawals to water availability* (Water Resources Vulnerability Index) has also been used in countries facing severe water scarcity (withdrawals exceed 40 per cent of the annual supply; Brown and Matlock 2011). Other indexes account for different uses of available water—for example, rain-fed agriculture (the *Water Poverty Index*; Salameh 2000) versus domestic and industrial uses of urban and rural population (Vörösmarty *et al.* 2005), which allows for partitioning water between differentiated users and supply sources. Arnell (2004) presented a country level assessment, at global scale, combining WSI at the catchment scale to quantify population exposed to water availability changes (improving or deteriorating) under future climate scenarios. They only produced aggregated data over the whole Mesoamerican region (comprising Central America and southern Mexico) without distinguishing between countries, whereas we seek here to produce regionally detailed estimates.

Methods

Study area

Central America comprises seven countries (from south to north, Panama, Costa Rica, Nicaragua, Honduras, El Salvador, Guatemala and Belize) that bridge South

and North America. It is a topographically complex strip of land where the Central American cordillera (reaching over 4000 masl) separates the Caribbean Sea and the Pacific Ocean by just over 60 km in some places. The region has a tropical climate with a dry season between December and May, when the rainy season begins until December with two peak months in June and September (Magaña *et al.* 1999). The region has a high inter-annual variability in precipitation leading to alternating periods of extreme rainfall regimes (Hastenrath and Polzin 2013).

Central America has approximately half (51 per cent in 2008) of its population (42.5 million in 2010) living in poverty and 26 per cent in extreme poverty with Costa Rica and Honduras showing the lowest (19 per cent) and highest (69 per cent) values respectively (CEPAL 2011). Water resources distribution is highly variable across the region and determined by climate and weather variability, human settlements, industry, and agricultural development (CEPAL 2011). Water demand is higher in the Pacific relative to the Caribbean watershed although the later has higher water availability (CEPAL 2011). The agricultural sector has the largest demand in Honduras, Guatemala, Costa Rica and El Salvador (between 54 per cent and 83 per cent of total water extracted) while in Panama and Belize extraction is dominated by the industrial and municipal sectors (66 per cent and 89 per cent respectively; CEPAL 2011).

The MAPSS hydrology and vegetation model

Hydrological ecosystem services, specifically water balance (i.e. runoff quantity) were evaluated for different climate change scenarios (under RCP 4.5) using a model that estimates the equilibrium between water balance and potential vegetation (Neilson 1995). The MAPSS model used in this study belongs to the group of soil–vegetation–atmosphere transfer (SVAT) models that are commonly used to simulate ecosystem functioning (i.e. Krinner *et al.* 2005) under changing climate conditions. A mechanism-based approach (such as the one used in MAPSS) is useful to assess changes in ecosystems types (Yates *et al.* 2000) and functions under changing environmental (e.g. climate change) conditions (Sitch *et al.* 2008). On the other hand, statistical models (i.e. Holdridge 1947) allow modeling vegetation based on the observed match between different climates and vegetation types but cannot account for the effects of new conditions (e.g. new climates or elevated CO_2) that have no current analog, which precisely will prevail over Central America (Williams *et al.* 2005). Approaches similar to the one used here have been used to assess changes vegetation type (Neilson 1995), ecosystems carbon stocks (Kindermann *et al.* 1996; Dargaville *et al.* 2002) and water cycles (Neilson 1995).

MAPSS simulates the vegetation distribution and structure (LAI) and precipitation partitioning (into runoff, soil moisture change and evapotranspiration) under a given climate (Neilson 1995). Potential vegetation cover in equilibrium with climate is modeled with a maximum LAI that can be supported based on soil texture, depth and climate input data (precipitation, temperature, wind speed and vapor pressure). Trees, shrubs and grasses compete for humidity and radiation and equilibrium conditions evaluated (for example, as the tree canopy closes grasses

disappear). It has been successfully used at high resolution for continental areas (a full model description is given by Neilson 1995).

MAPSS works at monthly time steps and calculates a leaf area index for each life form (trees, shrubs and grasses) and stomatal conductance to upscale transpiration of the ecosystem canopy and soil water dynamics (Neilson 1995). Precipitation is intercepted depending on total monthly precipitation and vegetation coverage (LAI). Through fall precipitation (reaching the soil layer) is divided into surface runoff or soil infiltration (depending if the soil is saturated or unsaturated). There are three soil layers where these processes occur, with grasses being able to transpire water from the top layer while trees and shrubs also from the intermediate layer. The third, and deep layer is used for base-flow that later becomes runoff. Potential evapotranspiration (PET) increases with LAI. While stomatal conductance decreases as soils get drier and as the atmospheric demand for water increases (PET). PET is based on an aerodynamic turbulent transfer model calibrated by Neilson (1995). The effect of elevated CO_2 can be evaluated by modifying water use efficiency of the vegetation (see Imbach *et al.* 2012 for an example in Mesoamerica).

The model is run in loops until the annual water balance of trees and shrubs and the monthly water balance for grasses reach equilibrium conditions, defined by the fractional coverage of each type of vegetation that maximizes transpiration on each grid point depleting most of the available water.

The advantage of MAPSS is that it accounts for feedbacks between changes in vegetation type and LAI on the soil water balance, which can produce non-linear equilibrium states, that could explain runoff changes under changing climate conditions in Mesoamerica (Imbach *et al.* 2012). The limitations of MAPSS are twofold. First, MAPSS does not account for aquifers water storage (∅s), assuming that this term is negligible at the annual scale. Second, MAPSS does not calculate explicitly the horizontal water flow within a catchment, as influenced by soil, climate and topography (i.e. Gómez-Delgado *et al.* 2011), but the limited data availability for our regional-scale assessment would make the use of such an explicit hydrology model impractical. Further, neglecting routing processes remains a good approximation in Central America, given the relatively small size of catchments in this region, and the monthly time steps used.

We also acknowledge that one limitation of our approach that we consider potential vegetation only, over a study area which has significant areas with pastures and agriculture (DeClerck *et al.* 2010). At the annual scale over this region, however, Imbach *et al.* (2010) found no model bias related to the cultivated fraction of each catchment, suggesting that current land use has no dramatic effect on simulated water availability for long term averages at the regional scale.

Climate change scenarios

We constructed regional climate change scenarios, using the reference climate data from the WorldClim 1.4 database (www.worldclim.org; Hijmans *et al.* 2005) at 30 seconds spatial resolution (~1 km^2), that provides monthly average precip-

itation, maximum and minimum temperature for the 1950-2000 period. Future climate scenarios are from CMIP5 for RCP 4.5 that accounts for an intermediate global radiative forcing (or emission scenarios). We used future scenarios from 19 GCMs from CMIP5.[1] Downscaled climatologies for each GCM was obtained as monthly 20-year averages for 2050 (2041–2060) and 2070 (2061–2080). The downscaling method is a simple approach known as the "delta method" where coarse resolution climate anomalies (GCM modeled difference between future and reference climate conditions) are added to the high resolution climatology (WorldClim 1.4 in our case).

Runoff and water availability

We estimated historical runoff based on the results from the above-mentioned MAPSS model calibrated by Imbach *et al.* (2010), as explained above, as well as future changes in water balance under average climate conditions in 2050 (2041–2060 average) and 2070 (2061–2080 average) under the studied scenario (RCP 4.5). Previous work from Imbach *et al.* (2012), using the same model setup, chose longer term climate scenarios (2070–2099) from a previous GCM dataset (CMIP3) and three scenarios (A2, A1B and B1).

We selected a runoff change threshold of 20 per cent in order to assess the likelihood of impacts across the range of climate scenarios (Imbach *et al.* 2012). The likelihood of change was estimated as the fraction of the 19 GCM+MAPSS model runs showing runoff change above 20 per cent (Mastrandrea *et al.* 2010). A change was considered as *likely* when runoff increased or decreased of at least 20 per cent in more than 66 per cent of the model runs. This procedure allowed showing both the magnitude of changes in runoff and the uncertainty from climate change scenarios.

We also calculated water availability at the catchment scale (300 catchments in total) using the WSI indicator (see the introduction to this chapter) with the catchment database from (Lehner *et al.* 2008; available from hydrosheds.cr.usgs.gov). This index was selected since the data for its estimation was available at the catchment scale covering the region of study. The WSI was calculated by aggregating total annual runoff for each catchment and its total population count from GRUMPv1 (Global Rural Urban Mapping Project; CIESIN *et al.* 2011). Mean runoff under future climate conditions was estimated as the median value from model runs using all the GCMs. We also excluded catchments smaller than 100 km^2 because these small watersheds often import water from neighboring larger catchments.

Results

Climate scenarios

Mean temperature anomalies for RCP 4.5 (average of the 19 GCMs) range between 2.2–1.4°C and 2.6–1.8°C for 2050 and 2070 respectively (1.8°C and

2.2°C mean values for the region shown in Figure 3.1a), with larger anomalies in northwest part of the region relative to the southern part of the study area.

Mean precipitation anomalies for 2050 for RCP 4.5 (average of 19 GCMs) show increased or decreased precipitation depending on the location considered in the Mesoamerican region. Most of the northern region (from Guatemala to Nicaragua) presents an average decrease of precipitation up to 10 per cent yr^{-1} while the southern countries (Costa Rica and Panama) show increased precipitation in similar magnitudes (Figure 3.1b). A precipitation increase occurs in the CMIP5 models over all four trimesters of the year in the southern part. In the northern part, mean annual precipitation decrease is explained by large decreases between June - August during the wet season, whereas small rainfall increases are observed during other months. A comparison of temperature and precipitation anomalies with previous studies based on AR4 scenarios and CMIP3 models (i.e. Imbach *et al.* 2012) is out of the scope of this study, given different radiative forcing trajectories between SRES (AR4) and RCP (AR5) scenarios, different number of GCM realizations and, different model versions. Mean annual anomalies of precipitation over the northern part of the region show relatively higher inter-model agreement relative to southern countries. We found *likely* negative precipitation anomalies for most of Guatemala, Honduras, Nicaragua (northern part) and El Salvador, and *likely* positive precipitation anomalies over Panama. Nicaragua and Costa Rica show negative and positive mean anomalies respectively, although with higher model disagreement on the anomaly sign (Figure 3.1b).

Population density and water balance

The least populated areas are usually located towards the Caribbean coast, including northern Guatemala, Belize, eastern Honduras and Nicaragua and eastern Panama. Belize and El Salvador are the least and most densely populated countries respectively (Figure 3.2). Areas with high population density are on the Pacific side, western Honduras and central areas of Costa Rica. Costa Rica and Panama have a relatively higher mean annual runoff (>1000 mm) across most of their territory (see Imbach *et al.* 2010) including areas with high population density. Contrastingly, Guatemala and Honduras have a large fraction of their most populated areas with lower water availability (mean annual runoff <500 mm).

We found that runoff is *likely* to decrease in 81 per cent of Central American countries in 2050 (89 per cent in 2070), following a general drying trend over the region (Figure 3.3). Runoff is *likely* to decrease by at least 20 per cent in over 50 per cent of the region in 2050 (71 per cent in 2070) and *likely* to increase over only 2 per cent of the area. Country per centages show that El Salvador has the largest area exposed to *likely* runoff reductions (of at least 20 per cent) followed by Belize, Costa Rica, Guatemala, Nicaragua, and Honduras (Table 3.1). Honduras has 33 per cent of its area exposed to *likely* runoff reductions. As expected, the areas of *likely* runoff reductions (>20 per cent) are larger in 2070 than in 2050, particularly for Honduras and Nicaragua (Table 3.1).

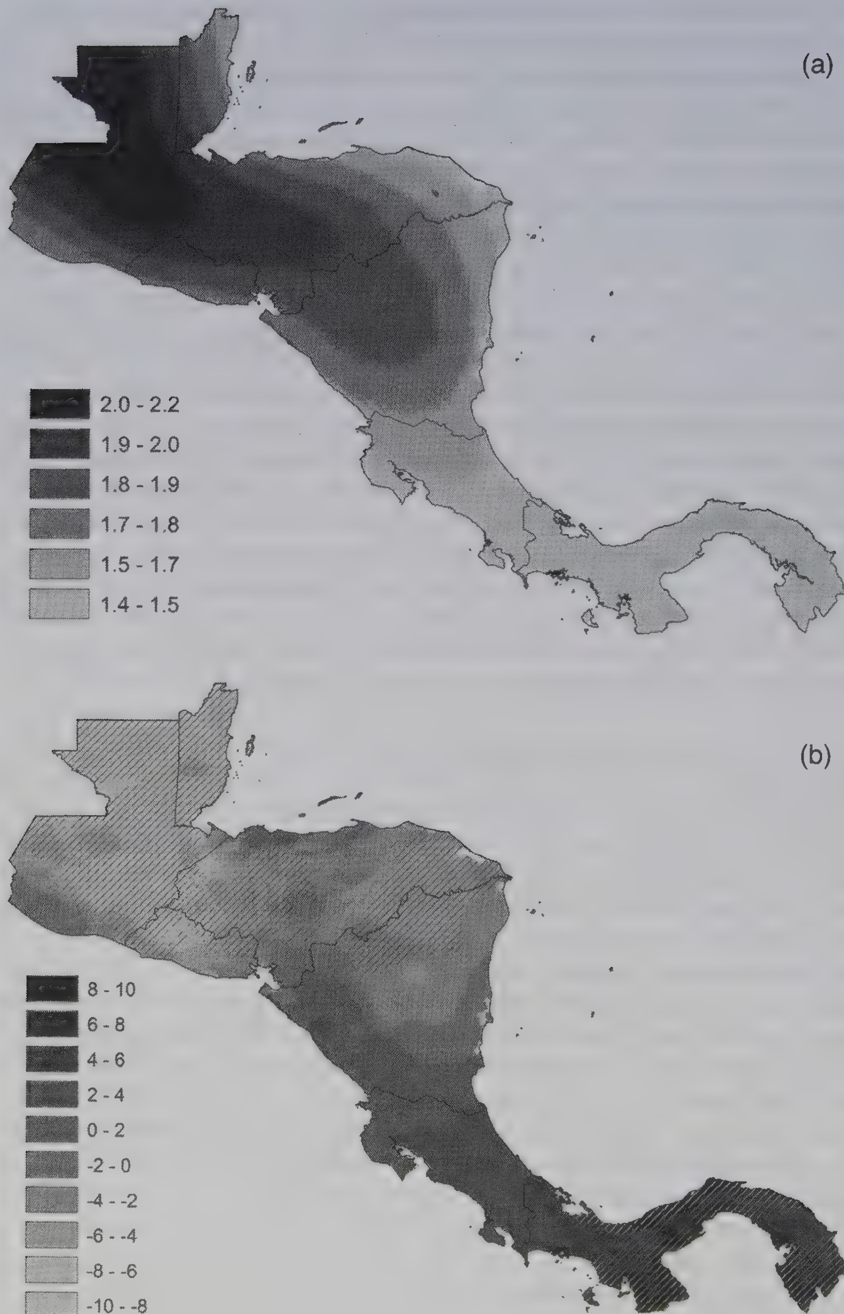

Figure 3.1 (a) Mean temperature and (b) precipitation anomaly (19 GCMs). Dashed areas show *likely* (>66%) model agreement on the anomaly sign

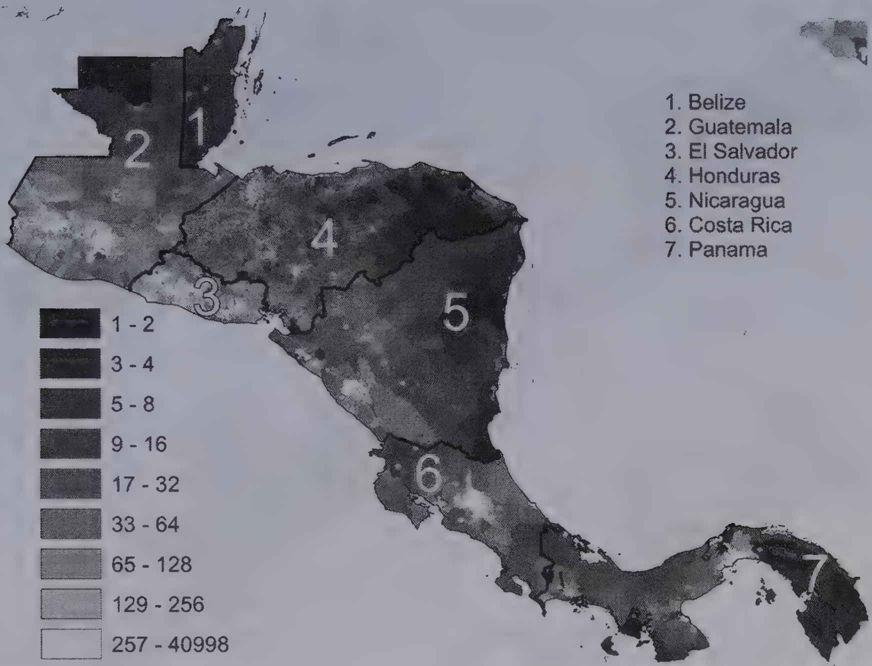

Figure 3.2 Population density (population count per 1 km^2 pixel)

Water availability under reference climate conditions (1950–2000) show *scarce* conditions over Honduras in Aguán, Cangrejal, Choluteca and Chamelecón (*absolute scarcity*) river basins. Guatemala (María Linda) and El Salvador (between Jiboa and Chilama river basins) show *stressed* basins. Nicaragua (between Tamarindo and Brito river basins) and Panamá (between San Juan Díaz and Pacora river basins) have both *scarce* and *stressed* basins near capital cities. Guatemala also shows limited resources within the Mopán and Hondo river basins (Figure 3.4a). Basins with limited resources hold 5.5 million people (around 15 per cent of the total population, including inhabitants in Mexico who share basins with Guatemala). Around 1 million people live in *absolute scarcity* conditions in the Chamelecón (Honduras) and Mopán-Hondo (Guatemala) river basins, although the later holds a lake that could improve resource availability (not accounted for in this study).

Current *scarce* resource availability remains unchanged under mid-twenty-first-century climate in Choluteca, parts of the Tamarindo–Brito and Juan Díaz–Pacora basins (in Nicaragua and Panama respectively). The Chamelecón basin persists under *absolute scarcity* and the Aguán and Cangrejal basins join this category in the future (Figure 3.4b). The Ulúa, Lempa, Banderas, Grande de Sonsonate, Paz and Atitlan river basins move from the *no stress* to the *stressed* category encompassing

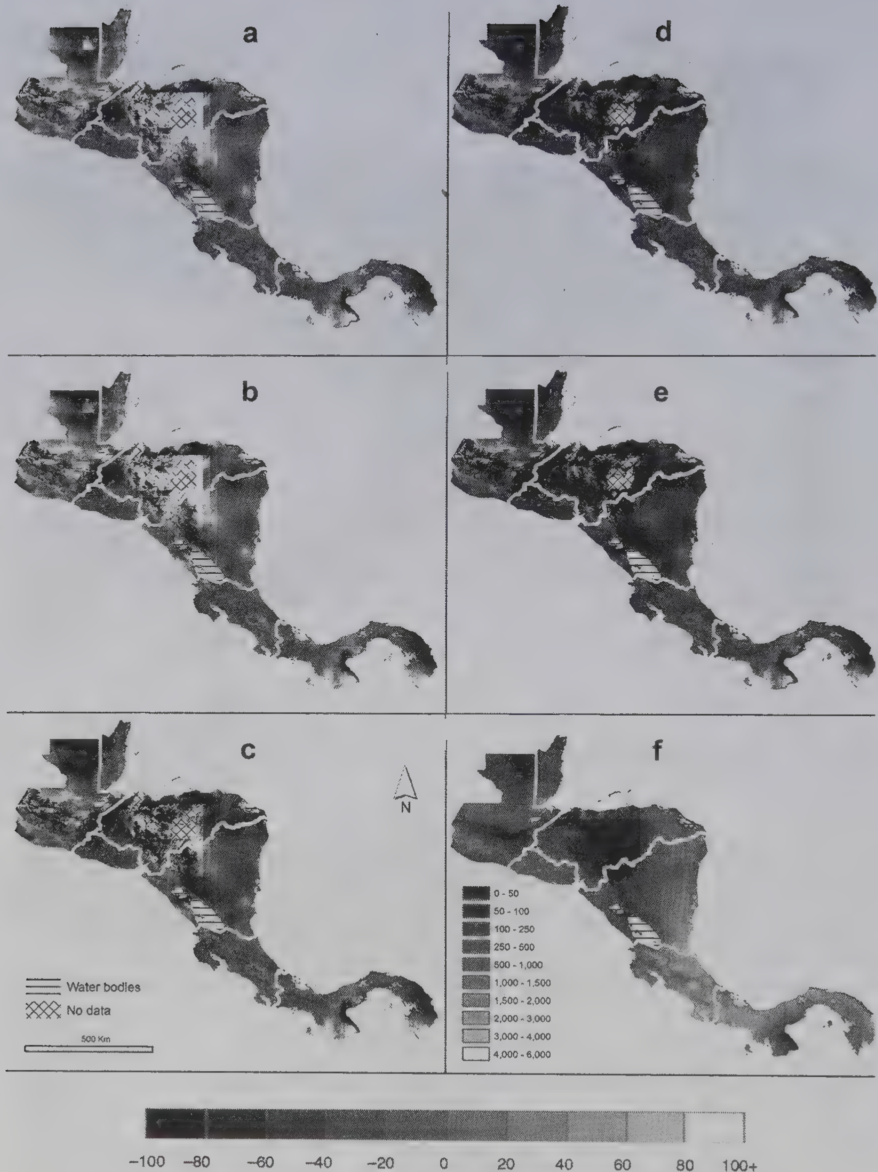

Figure 3.3 Change in annual runoff (%) for the (a) maximum, (e) minimum, (d) 25th, (c) 50th, and (b) 75th percentile values of the projected climate conditions in 2050 for the downscaled CMIP5 ensemble of 19 GCM models (all for the RCP 4.5 scenario), compared with (f) the reference period (1950–2000) (mm)

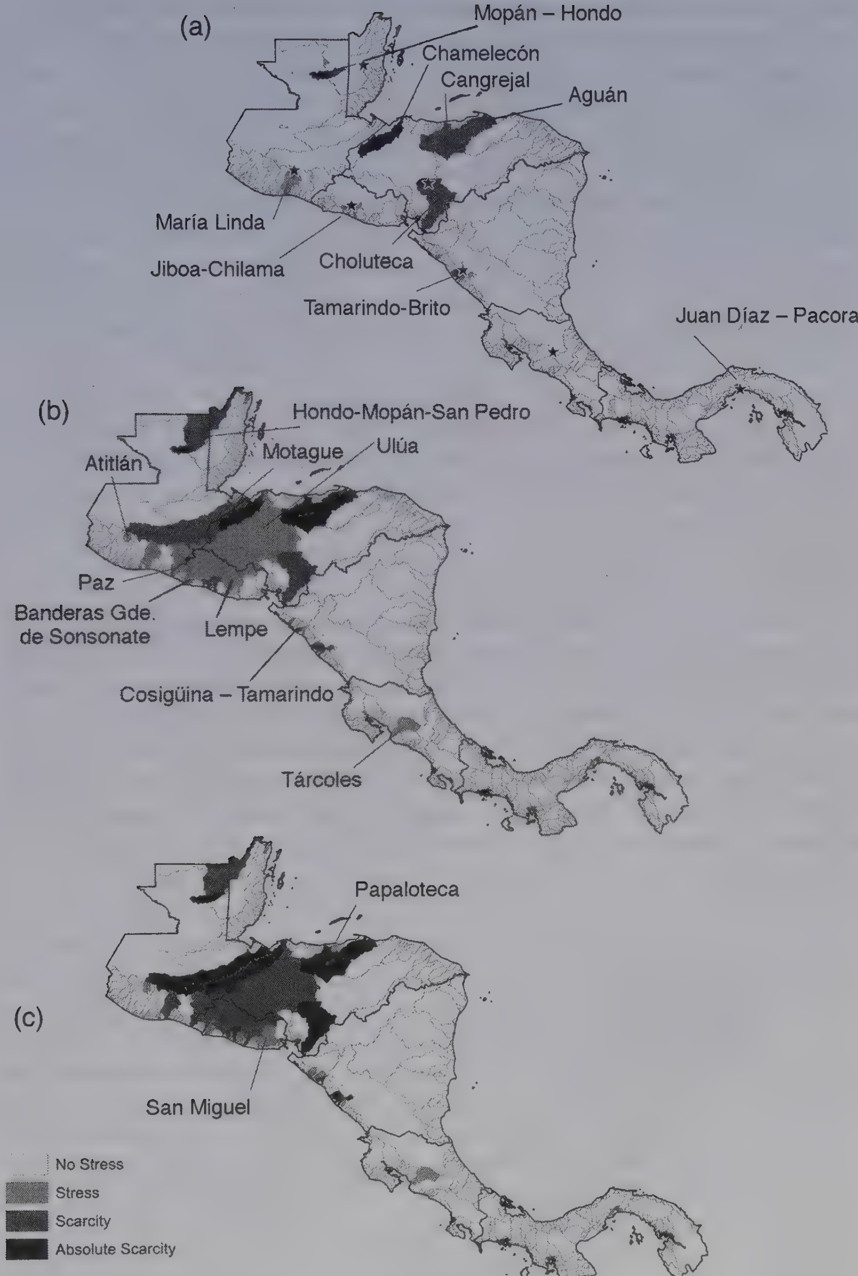

Figure 3.4 Water availability under (a) current and projected (b) 2050 and (c) 2070 climate conditions assuming the same population density as in year 2000. Water availability is measured in m^3/per capita where *absolute scarcity*, *scarcity*, *stress* and *no stress* thresholds are <500, 500–1000, 1000–1700 and >1700, respectively. Stars denote the location of capital cities

a large area with limited resource availability between Honduras, El Salvador and Guatemala. The Motagua basin also modeled to be under *scarce* condition by 2050. The Tárcoles basin in Costa Rica moves to the *stressed* category (from *no stress*) while basins with capital cities see their water availability worsened (Figure 3.4b).

For the 2070 period, the same pattern prevails but gets amplified (poor-get-poorer behavior) with a further decrease in water availability, except for the San Miguel and Papaloteca river basins that become *stressed* only in 2070 (Figure 3.4c). Under future climate conditions (2050 and 2070), we found an increase in the number of people living with limited water resources, with all areas falling into the stressed categories. A fraction of 36 per cent of the population currently living today without *stress* will fall into the limited resource availability category (10.9 million), and in fact mostly into the *stressed* category (with an increase of 6.9 million people in this category). Under 2070 climate conditions, the number of people with limited resources remains almost constant compared to 2050 (17.2 million) but the number living in *absolute scarcity* conditions is projected to increase by 4.8 million (from 2050 to 2070; Table 3.2).

Discussion

Our water balance modeling framework estimates changes of equilibrium conditions with historical and future climates, accounting for feedbacks between hydrological and vegetation dynamics that can result into complex and non-linear responses. Population density and land use are assumed to be constant in the future. Other hydrological modeling approaches evaluate effects of climate variability (i.e. seasonal and inter-annual) on water balance (Hidalgo *et al*. 2013) but usually use prescribed vegetation parameters and thus cannot account for the feedbacks of vegetation changes on runoff.

Table 3.1 Percentage area of Central American countries with *likely* runoff change (at least 20% decrease or increase) under 2050 and 2070 climate conditions (RCP 4.5) simulated by 19 GCMs of the CMIP5 ensemble

Country	*2050*		*2070*	
	Increase (>20%)	*Decrease (>20%)*	*Increase (>20%)*	*Decrease (>20%)*
Panama	0	43	0	55
Costa Rica	0	58	0	70
Nicaragua	0	50	0	82
Honduras	6	33	4	55
El Salvador	0	87	0	93
Guatemala	1	54	0	72
Belize	0	65	0	81

Table 3.2 Water availability for human use in Central America under current and future (2050 and 2070) climate conditions (people in millions/percentage). Water availability is defined in m^3/per capita where *absolute scarcity*, *scarcity*, *stress* and *no stress* thresholds are <500, 500–1000, 1000–1700 and >1700 m^3/per capita, respectively

Threshold	*Current*	*2050*	*2070*
No stress	30.6 / 85	19.7 / 55	18.8 / 52
Stress	2.3 / 6	9.2 / 26	3.0 / 8
Scarcity	2.2 / 6	5.4 / 15	7.8 / 21
Absolute scarcity	1.0 / 3	1.6 / 5	6.4 / 18

Although Imbach *et al.* (2012) performed a previous probabilistic water balance assessment for the region from the SRES climate scenarios of AR4 GCMs (CMIP3), we present here newer GCM simulations and shorter time-horizon scenarios (2050/2070) from AR5, and estimate water availability per capita for each GCM climate. The CMIP5 + MAPSS scenarios showed here show a similar uncertainty range of runoff change than the CMIP3 + MAPSS ones from Imbach *et al.* (2012), indicating a general decrease in river discharge across both previous and current generation of climate models and global warming scenarios. We found 70 per cent of the total area with *likely* decreases in 2070, similar to the results for year 2085 using CMIP3 models. Although, radiative forcing under RCP 4.5 in 2070 (3.84 $W.m^{-2}$) is relatively similar to SRES B1 in 2080 (4.09 $W.m^{-2}$), the smaller ensemble used in this study could have an effect on uncertainty estimates. The ensemble used in Imbach *et al.* (2012) was larger because it included different scenarios with the same GCMs (assuming equal weights for each one of them) their pattern is similar to those used in this study, with increased temperature and decreased precipitation for the northern part of the region and increased temperature and mixed signals (between) models for precipitation anomalies. As expected, shorter time horizons have smaller impacts, with half of the region with *likely* runoff reductions (>20 per cent) under 2050 climate conditions (compared to 70 per cent in 2070).

The approach presented here to estimate water availability has limitations that should be kept in mind for the interpretation of our results. First, our analysis is based on mean climate conditions and does not account for seasonal or inter-annual variability that could be important, particularly over dry areas where stress becomes important in dry years and/or with increased runoff variability (i.e. the Pacific watershed of Central America). This means, for example, that catchments found to be under current stress conditions can have sufficient runoff over specific years or within wet seasons. Other studies, at coarser scales, have accounted for inter-annual variability to estimate a significant change in runoff (where change in runoff is assumed significant only if larger than one standard deviation of inter-annual variability of a 30-year reference period; Arnell 2004). Second, this analysis uses the intermediate RCP pathway RCP4.5 and therefore does not explore potentially higher/lower climate forcing scenarios. Third, we estimate changes in

water availability assuming no changes in population or water resource demands (for example, for irrigation under drier future conditions). However, increase in water demand (accounting for population increase) has been estimated at 296–364 per cent by 2050 depending on the climate scenario (CEPAL 2011). Fourth, the model estimates water balance with potential vegetation and therefore assumes no changes in land use, which could have an effect at other temporal or spatial scales. Finally, and more importantly, we assumed no water transfer between catchments that could lead to a potential redistribution of resource availability, an issue that can be of relevance particularly in basins with large population centers that may develop infrastructure to access resources from other catchments or dam existing catchments to save water (e.g. for the dry season). This could be the case, for example, of all *stressed* catchments under current climate conditions on the Pacific watersheds containing capital cities that could have resources to transport water from nearby catchments.

The Chamelecón basin (Figure 3.4a–c, in Honduras) is the most water limited one under current conditions with availability close to the upper threshold of *absolute scarcity* (479 m^3 per capita), yet keeping in mind a runoff model underestimation of 13 per cent. Aguán, Cangrejal and Choluteca also show water *scarcity* under current conditions and the former two moving to *absolute scarcity* under future conditions in this study. CEDEX and SERNA (2002) estimated a hydrological balance of Honduras, based on water resources and sectorial demand estimates, and also found Chamelecón and Cangrejal basins among those with the smaller water surplus (while assuming no problems with access to the resource). They also found the Chamelecón River basin as the only case in Honduras with water deficit at the sub-basin scale (lower parts of the basin). The Hondo basin, in Guatemala, has a treaty dealing with international border issues and equitable use of the resources dating from 1961, indicating potential issues with resource availability (UNEP *et al.* 2007). CEPAL (2011) found El Salvador with *stress* conditions under current climate using a long term average model for water availability and differentiating water use by different sectors (i.e. human consumptions, agriculture and industrial). Changes in future sectors demand were also assessed by the CEPAL study, and are assumed constant as in our case, although their study assessed changes that did not account for within-country disparities in resource availability (only the total country-level resource availability) as well as resources available from trans-boundary basins.

We found a general trend of increased water scarcity in the northern part of Central America, mostly over Guatemala, Honduras and El Salvador, under future scenarios (under RCP 4.5). Although the region uses around 8 per cent of available water resources currently, country rates of use are variable. For example Costa Rica and Nicaragua use 20 per cent and 1 per cent of their total available resources respectively today (GWP *et al.* 2011), meaning that reduction trends will have variable impacts. Furthermore, several basins are shared with neighboring countries (international basins cover 37 per cent of the region, or 191,449 km^2), and these watersheds cover between 75 and 5 per cent of country territories (for Guatemala and Panama respectively showing the largest and

smallest per centages; GWP *et al.* 2011). Internationally shared basins have been the source of conflicts in the past (i.e. Lempa river basin linked to siltation problems; Wolf 2007), where decreasing resource availability could become an important factor.

Box 3.1 From emission scenarios to representative concentration pathways

The assessment presented on the fourth IPCC report (AR4; IPCC 2007) was based on simulations from global climate models from the Coupled Model Intercomparison Project 3 (CMIP3) forced by green-house gases (GHG) emissions scenarios from the Special Report on Emission Scenarios (SRES; IPCC 2000). SRES emission scenarios were derived from integrated assessment models (models that explore the physical, biological, economic and social components for impact and policy response assessments) of specific future storylines of demographic and economic development, energy use, technology and land use (IPCC 2000). IPCC fifth assessment report (AR5; IPCC 2013) uses global climate simulations from the Coupled Model Intercomparison Project 5 (CMIP5) forced by representative concentration pathways (RCP). The RCPs are scenarios depicting possible future trajectories of future emissions and concentration of greenhouse gases, air pollutants and land use change covering the range of future radiative forcing (a measure of the global net radiation imbalance at the top of the atmosphere, and measured in $W.m^{-2}$, that is directly related to global mean temperature) found in the literature but that are not based on specific socio-economic storylines (Cubasch *et al.* 2013). RCPs can be explored by the community developing integrated assessment models in order to develop socio-economic and policy storylines while, in parallel, climate modelers develop future climate scenarios (Vuuren *et al.* 2011). The comparison of RCP and SRES based scenarios is rendered complex due to the different emissions scenarios and climate models used for each assessment report. Rogelj *et al.* (2012) however provide a comparison between both (AR4 and 5) climate scenarios based on a common analysis framework of equilibrium climate sensitivities. They found that the range of changes in future mean global temperature is larger for RCPs (likely changes of 1.3–5.7°C for 2090–2099) than SRES (2.0–5.8°C) and differences between specific scenarios. For example, RCP 4.5 global median temperatures (likely changes of 2.0–2.9°C for 2090–2099) have a faster increase until 2050 and slower afterwards when compared to SRES B1 scenario (2.0–3.1°C). RCP 8.5 global median temperatures (likely changes of 3.8–5.7°C for 2090–2099) have a slower increase for the period 2035-2080 and faster in any other period when compared to A1FI (3.9–5.8°C; Rogelj *et al.* 2012). RCP 2.6 has a lower radiative forcing than any SRES scenario (Cubasch *et al.* 2013).

UNEP *et al.* (2007) assessed water availability in 2025 for international basins in Central America, where only the Choluteca basin appears *stressed* and the Ulúa, Lempa, Grande de Sonsonate, Paz, and Motagua (with limited resources in our results for 2050) appear just over the 1700 m^3 per capita *stress* threshold (using CGCM1GSa1 and HadCM2GSa1) under 1 per cent per year increase in CO_2 equivalent and sulfate aerosol dampening (Vörösmarty 2000). It is worth noting that this study used a coarser-scale hydrological model (0.5°~50 km; Fekete *et al.* 1999) that might miss some of the runoff variability. Although our study shows stress for the Atitlan and Mopán–Hondo basins, these hold large reservoirs that could allow for resource storage not accounted for in our study. Some basins move out the *no stress* category only under 2070 climate (San Miguel and Papaloteca).

Wolf (2007) estimates that although no Central American country has limited resources (based on mean national estimates), except for El Salvador, the isthmus has drinking water shortages due to accessibility issues. This study also highlights the fact that several capital cities (for El Salvador, Nicaragua, Honduras, and Belize) lay on international basins, where we found reduced future availability due to climate change. Furthermore, San Salvador (Lempa river basin) being currently the capital with most concerns by poor management and limited water resources availability.

Three quarters of the population living on the Pacific side rely mostly on aquifer resources CEPAL (2011) and several capital cities rely to significant extents on aquifer water resources. For example, Negro, Chixoy, La Vaca (a tributary of the Motagua river) in Guatemala; Choluteca in Honduras (5 per cent of urban demand from underground sources); Lempa in El Salvador (37 per cent, who are also used to generate 41 per cent of the country energy supply; Wolf 2007). These areas could have delays in water availability problems depending on water use and storage rates since large amounts of the resources could take longer times to be depleted (as compared to surface resources).

The Lempa River basin (among several other basins), in El Salvador and Honduras, has also suffered agricultural loses during the 2000–2001 droughts, also highlighting the role of climate variability in determining the impacts of water availability.

Arnell (2004) estimated in their global coarse-scale study that 16 per cent of Mesoamerican population lived under water stress (defined as less than 1000 m^3 per capita) in 1995, although their study area includes Central America and southern Mexico (the latter is excluded in our study). They also found a large number of people exposed to increased water stress depending on emission scenario and climate model (4–108 million) for Mesoamerica in 2025. Our results are on the lower part of his range, even when accounting for all people under stress (17.4 million under 2070 climate). Comparison of the two results findings is uncertain because Arnell (2004) used a range from specific GCM simulations and emission scenarios, while our approach is based on mean values of runoff (and for a smaller area or at higher resolution). This approach allows for reduced uncertainty in our estimates given the relatively clear signal of change in runoff for our study region. Some models (in Arnell's 2004 study) also show population that will experience

a decrease in water stress (between 0 and 82 million people), contrary to our results, although the differences could be areas within Mesoamerica that are outside of our study area.

Finally, it is important to note that our hydrological modeling approach is based on equilibrium conditions with climate and therefore does not account for transient changes meaning that we have no indication on when the evaluated impacts could happen since ecosystems and water dynamics could happen at a different pace than climate change, for example, changes in leaf area index could occur faster than changes in tree fraction cover (Jones *et al.* 2009).

Box 3.2 Policy implications

Given the threats of climate change to water resources and hydrological ecosystem services in Central America, adaptation is needed for all socioeconomic sectors depending on water and watershed services. For example, hydroelectricity is a major source of energy in this region, it is vulnerable to climate change and depends largely on ecosystem services for soil erosion reduction and water flow regulation (Locatelli *et al.* 2010). Flooding and drought are regularly affecting communities and economic activities (Simms and Reid 2006; Wunder and Wertz-Kanounnikoff 2009). Adaptation policies are emerging in all Central American countries but their design is made difficult by uncertainties about climate change trends and a cascade of unknowns about biophysical impacts and socioeconomic consequences. This impact assessment study provides scientific inputs from a biophysical perspective for adaptation options and the next step is to understand the socioeconomic implications and the policy context at regional, national and local levels. Because uncertainties inherent to impact assessments may be difficult to handle by policymakers (Burton *et al.* 2002), long-term and regional impact assessments must be complemented with vulnerability assessments aiming at understanding how and why people and water-dependent economic sectors are vulnerable to climate variations at the local level now and in the future.

Historically, water management policy decisions do not account for climatic change because climate was assumed to be unchanging. Decisions with short-term implications can reasonably ignore climate change and its uncertainties but this is not the case of water management plans, which require infrastructure, land use or socio-economic development in the long term. The approach used in our study helps acknowledging uncertainties, which is an important step in adaptation policy development (Dessai and Wilby 2011). Recognizing uncertainties about future climate change impacts calls for diversified and flexible approaches to adaptation. Depending on the local context, these approaches must combine different measures selected from an adaptation toolbox (i.e. a list of possible measures; Locatelli *et al.* 2008). Adaptation measures in the water sector

can involve the demand side (e.g. improving water use efficiency for reducing consumption), the transformation side (e.g. improving treatment and transportation infrastructure) and the supply side (e.g. managing watersheds). The selection of adaptation measures depends on their cost or feasibility but also on the outcomes that the society considers of interest, recognizing trade-offs between them. For example, landscape management in Central America can contribute to protect watershed services, biodiversity and other services (such as carbon or scenic beauty) at the same time, but trade-offs can occur (Locatelli *et al.* 2013). Flexible approaches also require monitoring outcomes and learning from experience in order to achieve an adaptive management.

Because of the complex relationships between climate, ecosystems and water and the role of ecosystems in regulating water quality and quantity, adaptation plans in the water sector cannot be limited to brick and mortar solutions. Central America has a long experience of watershed management and payment for environmental services (PES), particularly in relation with water (Kaimowitz 2005; Kosoy *et al.* 2007). Recent studies and reviews have highlighted that ecosystem management can contribute to the adaptation of the society to climate change (the so-called ecosystem-based adaptation; Pramova *et al.* 2012) and that instruments such as PES can be tailored to become adaptation instruments (Wertz-Kanounnikoff *et al.* 2011). But to conserve ecosystem services that are important to help society to adapt to climate change, adaptation measures must also be designed for those ecosystems, for example through reducing human pressures, conserving biodiversity hotspots and improving landscape connectivity between protected areas (Guariguata *et al.* 2008). For these reasons, the Mesoamerican Biological Corridor has an important role to play in ecosystem adaptation and, thus, societal adaptation (Imbach *et al.* 2013).

The following is an example of interesting initiatives that are emerging in Central America on climate change adaptation. In September 2010, the Adaptation Fund of the United Nations Framework Convention on Climate Change accepted its first two projects. One of these, in Honduras, aims at improving water management and water security for the poor in capital region of Tegucigalpa (Locatelli *et al.* 2011). In addition to measures on the water demand and infrastructure sides, this project highlights the role of forests, for example how cloud forests capture mist from the atmosphere and how deforestation affects water supply. The project developers recognize that ecosystem management (including the creation of protected areas) is crucial for Tegucigalpa water supply and that there are currently no mechanisms to conserve hydrological ecosystem services (Adaptation Fund 2010). In addition to considering ecosystems for societal adaptation, the project also plans to implement adaptation for ecosystems: biological corridors will be conserved and restored to increase connectivity and facilitate ecological adaptation. This example shows how adaptation for people, water, and ecosystems can be integrated in a cross-sectorial approach.

Conclusions

We evaluated the impacts of climate change on water balance and per capita water availability in Central America watersheds using recent GCM results from IPCC AR5. We used a soil–vegetation–atmosphere transfer model, previously calibrated for our study area, to estimate mean runoff at 1 km^2 resolution under historical (1950–2000) and future (2040–2060 and 2060–2080) climate conditions for the intermediate global warming scenario RCP 4.5. We estimated per capita water availability based on population count per watershed and an index of water stress to assess changes in future resource availability. We found a general decreasing trend in water availability per capita, with resource availability limitations mostly in the northern part of Central America as well as in basins with high population density (i.e. capital cities). Our study updated previous water balance scenarios developed for the region- and watershed-scale indicators of potential stress in resource availability due to climate change.

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Note

1 ACCESS1-0, BCC-CSM1-1, CCSM4, CESM1-CAM5-1-FV2, CNRM-CM5, GFDL-CM3, GFDL-ESM2G, GISS-E2-R, HadGEM2-AO, HadGEM2-CC, HadGEM2-ES, INMCM4, IPSL-CM5A-LR, MIROC-ESM-CHEM, MIROC-ESM, MIROC5, MPI-ESM-LR, MRI-CGCM3, and NorESM1-M.

4 Climate change impacts on water services in Costa Rica

A production function for the hydro-energy sector

Elisa Sainz de Murieta, Aline Chiabai and Juan Carlos Zamora

Introduction

The impact of climate change on biodiversity and ecosystem services is widely documented in the scientific literature. The Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report asseverated that there is sufficient evidence of the changes in physical and biological systems already happening due to global warming (IPCC 2014), and even greater impacts on biodiversity are expected in the future (Thomas *et al.* 2004; Thuiller 2007). In fact, the Millennium Ecosystem Assessment report (MEA 2005) stated that climate change is expected to be the main cause for biodiversity loss and changes in ecosystem services at a global scale by the end of the twenty-first century. If global temperature rose more than 2–3°C above pre-industrial levels, 20 to 30 per cent of species will be at high risk of extinction and important changes in the structure and function of terrestrial ecosystems are very likely to occur (Fischlin *et al.* 2007). The changes provoked by climate change add to previous anthropogenic pressures on the ecosystems and the services they provide, such as deforestation, land-use changes, habitat loss or the overexploitation of natural resources (MEA 2005).

Climate change is characterized by an increase in global average temperature resulting in a range of changes in mean temperature and precipitation, and its extremes across continents and hemispheres. In the case of the Central American region, future scenarios indicate a general drying trend, due to an increase in temperature and a reduction of total annual rainfall (Neelin *et al.* 2006). This drying trend is expected to have a strong effect on water resources and hydrological ecosystem services. According to Arnell *et al.* (2004), the number of people living in water-stressed watersheds in Central America will significantly increase due to climate change, with over 10 million people potentially moving into water-stressed conditions (as described in Chapter 3). Runoff is foreseen to be reduced in all of the region regardless of future scenarios and potential vegetation will shift from humid to dry types, even in areas where precipitation will potentially increase, as the rise of temperature will intensify evapotranspiration reducing water availability (Imbach *et al.* 2012). In summary, the expected changes indicate that Mesoamerica

is a climate change hotspot among tropical regions (Giorgi 2006). In the case of Costa Rica, projections carried out by the Ministry of the Environment (Ministerio de Ambiente, Energia y Telecomunicaciones 2009) for IPPC scenario A2 show an increase in mean temperature, while annual precipitation trends varies among regions. Regarding runoff and evapotranspiration, Imbach *et al.* (2012) estimated a reduction of annual runoff and a likely increase of evapotranspiration by more than 20 per cent.

Climate change will not only affect the physical and natural systems, but it could also hamper socioeconomic development. This study focuses on the energy sector, which is a major component of development but also an important contributor to greenhouse gas (GHG) emissions. Nearly 64 per cent of global GHG emissions related to human activities come from the energy sector (Emberson *et al.* 2012). At the same time, this sector is vulnerable to the impacts of global change. Energy supply and demand, energy endowment, infrastructure, and transportation could be directly affected by climate change along with changes in economic or natural systems (Ansuategi 2014). For example, water resources are strongly linked to some energy sources, where climate change impacts on water production, availability and quality may have an effect on the energy production (Haas 2009). This is the case of the hydroelectric sector, as the amount of electricity produced is determined by the installed generation capacity, but also by the water influx to the hydropower plant (Schaeffer *et al.* 2012). Hydropower production is, therefore, highly dependent on climate variability and hydrologic ecosystem services, such as water quantity regulation, sediment retention and reduction of soil erosion (Kaimowitz 2004).

The study presented in this chapter was conducted in Costa Rica, where major economic sectors, such as agriculture and energy, strongly depend on water resources. The hydroelectric sector plays an important role: in 2013, the installed hydropower capacity was 1725 MW, which represents 63 per cent of the total installed capacity and 68 per cent of the total energy produced that year in Costa Rica (CEPAL 2013).

The aim of this chapter is to estimate the economic value of water production services for hydropower and to project the expected changes in hydro-energy production under different climate change scenarios in Costa Rica, by 2100. For this purpose, an innovative multidisciplinary approach is used combining bio-physical, technical and economic data. The model is based on a production function relating mean annual runoff, estimated by a soil–vegetation–atmosphere transfer (SVAT) model (Imbach *et al.* 2010), with changes in energy production for 35 hydropower plants in Costa Rica and other explanatory variables deemed to have an influence on the production of energy. Also, this methodology can be easily adapted to be applied at the regional scale taking into account the existing hydropower plants in Central America and doing the analysis by watershed unit.

The conceptual framework is explained next, after which the methodology and data used are described in detail. We then show the empirical application, and discuss results and policy implications, before presenting the main conclusions.

Conceptual framework

The Millennium Ecosystem Assessment (MEA) was launched in the year 2000 seeking to evaluate the "consequences of ecosystem change for human wellbeing and to set up the scientific basis for actions needed to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being" (MEA 2005). Ecosystem services (ES) support human welfare at different scales either directly or indirectly, from climate regulation and carbon sequestration occurring at the global scale, to flood protection, soil formation, erosion prevention and nutrient cycling occurring at the local and regional scales (MEA 2005).

The MEA classifies ecosystem services into four groups: provisioning services such as food, water and timber; regulating services that affect climate, floods, or water quality; cultural services that provide recreational, aesthetic, and spiritual benefits; and supporting services such as soil formation and nutrient cycling. In this conceptual framework people are integral parts of ecosystems in a way that a dynamic interaction exists between them and other parts of ecosystems. Therefore, human activities directly and indirectly alter the ecosystems and, in turn, changes in ecosystems have an effect on human wellbeing (see Chapter 2 for more details).

The conceptual framework of this study is based on the approach defined by Brauman *et al.* (2007), who by taking the MEA framework and ecosystem classification as a basis define a specific way for assessing hydrologic services. According to the authors, hydrologic ecosystem services are the benefits provided to people by "terrestrial ecosystem effects on freshwater" (Brauman *et al.* 2007: 72). These services represent a wide group of benefits that the authors divide into five categories:

- extractive water supply,
- *in situ* water supply,
- water damage mitigation,
- spiritual and aesthetic, and
- supporting services.

These five kinds of benefits that the beneficiaries receive are determined by water quantity and quality, location and flow timing (what the authors call "hydrologic attributes"), which in turn depend on the eco-hydrologic process performed by the ecosystem. In summary, every ecosystem in a watershed will affect one or several hydrologic attributes. Through this effect on the hydrologic attributes, we can say that the ecosystems define and regulate the hydrologic services (Figure 4.1). The conceptual framework of this study is based on the approach defined by Brauman *et al.* (2007) as it is an output-based classification, where runoff is the hydrological attribute and hydro-energy production falls in the category "in situ water supply." This perspective is perfectly aligned with the ideas behind the production function, as discussed below. Obviously, the relationship between the ecosystems, the water or hydrological attributes and the services provided is not

simple and trade-offs occur. The services can compete with each other and some services will improve at the expense of others. For example, a great quantity of water could be very positive for hydropower, while negative for flood prevention (Brauman *et al.* 2007).

In this study, we consider forest ecosystems as provider of a variety of hydrological services, such as flood control, groundwater recharge, water quality improvement and soil retention (Lele 2009). The hydrologic service considered is the provision of water for hydropower generation and the attribute examined is the volume of total annual runoff (quantity) at the catchments as a proxy for the water volume used for hydro-power generation. Mean annual cycles of runoff were modelled using the SVAT model MAPSS (Imbach *et al.* 2010). The unit of analysis is the watershed, since it represents the draining area for the water used in each hydro-power plant.

Other services provided by forest ecosystems, which are equally important for hydropower production, such as soil retention and water quality enhancement, were not included in the analysis due to lack of data.

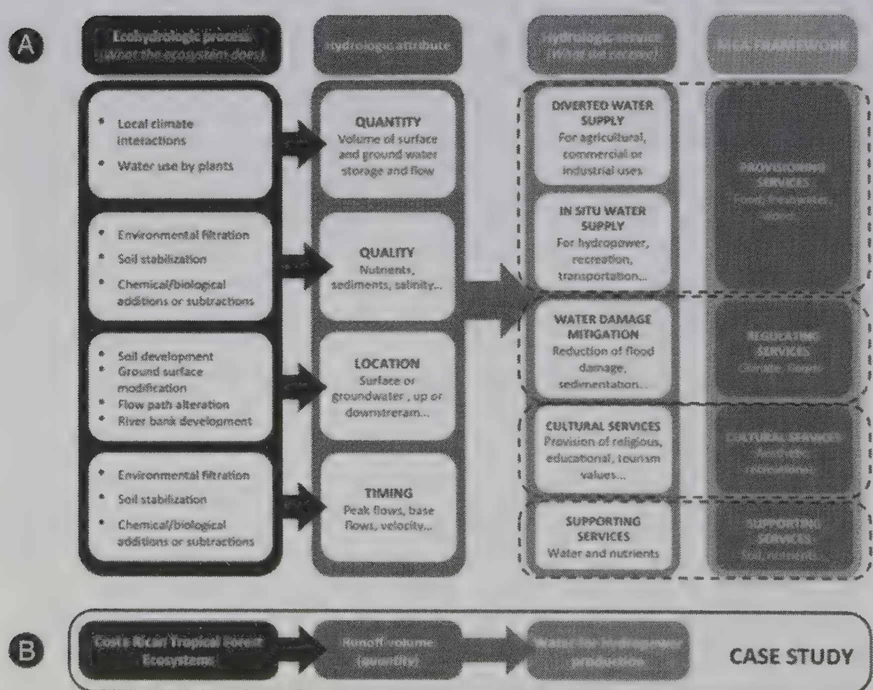


Figure 4.1 (A) Relationship between MEA general framework and Brauman approach, which shows the complex connection between hydrologic ecosystem services and processes. (B) The Brauman framework applied to the case study

Source: adapted from Brauman *et al.* (2007) and MEA (2005).

A physically driven economic valuation methodology

The economic valuation of ecosystems services allows, first, to provide a monetary estimate in terms of specific benefits generated to humans, and second, to estimate the economic impacts on human activities on the basis of the damage produced on ecosystems and related services. However, economic valuation is not without criticism. Some argue that biodiversity has a value in itself, not just to the extent that it provides services to humans. Others believe that the monetization of biodiversity only considers a part of its total value, so that the result underestimates its true value (Markandya *et al.* 2008). Some studies include "conceptual errors, oversimplified biophysical models or lack of social and technological context" (Lele 2009). All these are appreciations that we share, but still we think that economic valuation can be a powerful tool when deciding about public policies that affect biodiversity and ecosystem services, as it allows putting a monetary value on expected impacts and making appropriate decisions in a world of scarce resources.

The economic value of ecosystem services can be estimated from two sources. First, ecosystem services can be measured as an input to the production of a marketed good or service. Second, they can be assessed for their direct contribution to human wellbeing, for example the pleasure of enjoying nature (Markandya *et al.* 2008). In this chapter, a production function is defined to estimate how the change in the provision of water, as input for the production of hydroelectricity, affects the economic activity of producing energy. This technique requires a multidisciplinary approach, so that the economic valuation is nourished by the results of a complex biophysical model that provides the hydrological information. More specifically, the proposed approach combines water balance outputs (in terms of surface runoff) from a SVAT model (Imbach *et al.* 2010) with electricity generation and economic revenues in a production function. The SVAT model MAPPS accounts for feedbacks between potential vegetation cover and water balance, and allows assessing climate change impacts on runoff. In the subsequent step the production function estimates the impacts that changes in annual runoff produce in the generation of hydroelectricity in terms of the revenues of the hydropower plants.

The production function

Background

The production function method, also known as the change in productivity method, "relies on the fact that ecosystems may be inputs into the production of other goods or services that are themselves marketed" (Barbier 2007: 178), such as electricity and agricultural products (see Chapter 2). Barbier (2000, 2007) valued respectively mangrove ecosystems and coastal wetlands in terms of the economic benefits provided to fisheries. In these studies, the production function is combined with dynamic models and intertemporal functions because fisheries are renewable resources and their growth rate is affected by ecological functioning.

Pattanayak and Kramer (2001) also used a production function to evaluate the drought mitigation services provided by the Ruteng Park protected area in Eastern Indonesia as benefits to agricultural households. The same approach was followed by Ricketts *et al.* (2004) to measure the value of tropical forests in supplying pollination services to coffee production in Costa Rica.

In this chapter we use a production function to estimate the economic value of runoff, which is a non-marketed good associated to electricity production. The methodology differs slightly from Barbier (2000, 2007) because the production function is not a dynamic one as the resource valued (hydro-energy production) is of different nature.

Our approach is based on that developed by Núñez *et al.* (2006) who measured the service provided by the native temperate rainforests in the Llancahue watershed (Chile). The objective of their study was to estimate the economic impacts of the conversion of native forest to plantations, in terms of the volume of stream water as the ecosystem input factor. This way, the authors constructed a production function for drinkable water, which allowed them to measure how the changes in water stream due to the substitution of native forests by plantations affected the production of drinkable water. Then, the changes in the production of drinkable water were measured through the production function and the economic value per cubic meter of stream water was calculated as the product of the price of drinking water and the marginal physical production of stream water.

The model

The implementation of the model implies two steps. The first step is to calculate the change in runoff for each of the watersheds were hydropower plants are located under different climate change scenarios; the second is to value the changes in runoff in terms of the corresponding change in the marketed good, that is, in our case, hydro-energy production. In this approach, runoff is considered an input to the hydropower activity: changes in runoff will cause changes in the production of electricity. Therefore, its value can be estimated through changes in the productivity of the hydroelectric plants considered (Barbier 2000; Núñez *et al.* 2006).

The impacts of climate change on water balance (surface runoff)

Water balance assessment under current and future climate scenarios was performed using the MAPSS (Mapped Atmosphere Plant Soil System) model. MAPSS is a SVAT model, first developed by Neilson (1995) and later calibrated and validated for Mesoamerica by Imbach *et al.* (2010). SVAT models are used to simulate ecosystems and their functioning under historical and future climate conditions. Their mechanistic approach is useful to assess changes in ecosystems structure and function under changing environmental (e.g. climate change) conditions. MAPSS simulates precipitation partitioning into runoff and evapo-

transpiration under long-term mean climate conditions. The model uses climate (monthly precipitation, temperature, wind speed and water vapor deficit) and soil data (texture, depth and rock content) to estimate water balance (runoff and evapotranspiration) and vegetation cover (leaf area index of grass, shrubs and trees). The methodology is explained in depth in Chapter 3, although the scenarios used in this chapter are those from the IPCC Fourth Assessment Report.

Baseline runoff was estimated using a high resolution dataset for mean climate conditions for the 1950–2000 period (Hijmans *et al.* 2005; available at worldclim.org). An ensemble of simulations from 23 general circulation models (GCM), under high (A2), intermediate (A1B), and low (B1)[1] emission scenarios, were used for the period 2070–2099, corresponding to the Coupled Model Intercomparison Project (CMIP3). The downscaled climate scenarios were calculated using the change factor method, where the coarse scale anomaly from the GCM output is added to high resolution climatology (Hijmans *et al.* 2005). Future changes in runoff (Imbach *et al.* 2012) were estimated using a model version calibrated for Mesoamerica (Imbach *et al.* 2010) and results are shown in Figure 4.2. Long-term mean annual runoff was calculated for 35 hydropower plants in Costa Rica whose catchment areas were delineated based on digital elevation data (see Figure 4.3).

Measuring the change in productivity

The productivity method was selected because this is a straightforward case where a hydrologic attribute (runoff quantity) directly affects the quantity of a marketed good—hydroelectricity, and therefore its revenues. Hydro-energy is characterized

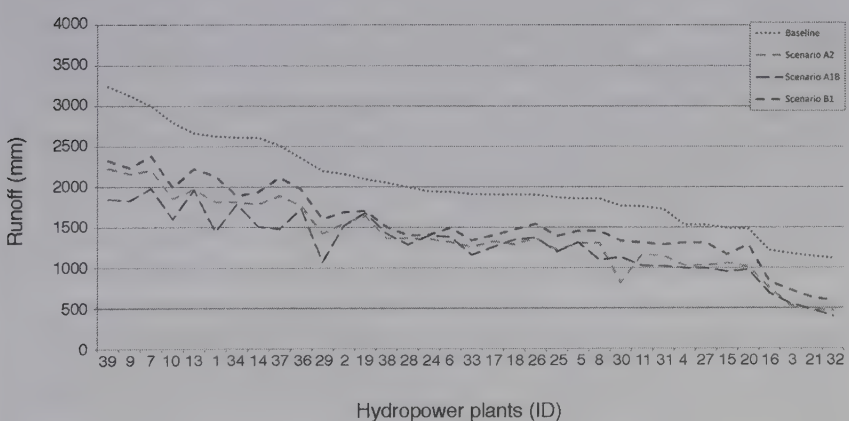
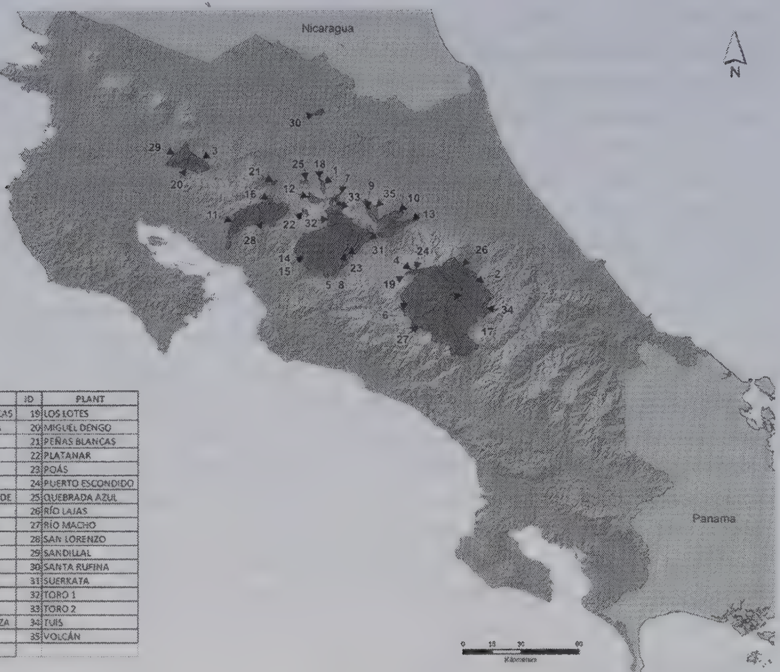


Figure 4.2 Estimation of annual runoff (mm) for each hydropower plant and for baseline and future scenarios. Plants are ordered from left to right according to the quantity of runoff produced in the corresponding watershed (note also that each point is independent and the figure does not show a flow)



ID	PLANT	ID	PLANT
1	AGUAS ZARCAS	19	LOS LOTES
2	ANGOSTURA	20	MIGUEL DENGO
3	ARENAL	21	PEÑAS BLANCAS
4	AVANCE	22	PLATANAR
5	CACAO	23	POÁS
6	CACHI	24	PUERTO ESCONDIDO
7	CAÑO GRANDE	25	QUEBRADA AZUL
8	CARRILLOS	26	RÍO LAJAS
9	DON PEDRO	27	RÍO MACHO
10	DOÑA JULIA	28	SAN LORENZO
11	ECHANDI	29	SANDILLAL
12	EL EMBALSE	30	SANTA RUFINA
13	EL GENERAL	31	SUERKATA
14	GARITA 1	32	TORO 1
15	GARITA 3	33	TORO 2
16	LA ESPERANZA	34	TUIS
17	LA JOYA	35	VOLCÁN
18	LA REBECA		

Figure 4.3 Watersheds where the studied hydropower plants are located

by a high capital inversion but low operational costs, and electricity sales are the only source to recover the inversion and generate revenues (Harrison and Whittington 2001). Thus, an increase or reduction in runoff from the watershed feeding the hydropower plant will have a direct influence on the energy production and therefore, on the revenues of the hydroelectric station.

Hence, the production function can be expressed as a mathematical function where the hydrologic attribute (runoff) relates to the service (hydroelectricity production):

$$Q = Q(X_i, \dots X_n, W) \tag{4.1}$$

where Q is the produced electricity or the revenue of the hydropower plant ($\$_{2009}$); X are the production factors and W is the hydrologic attribute (runoff).

If we consider a linear production function for equation 4.1 and we add several production factors, the function is defined by:

$$Q = \alpha + \beta_1 X_1 + \dots \beta_n X_n + \beta_w W + u \tag{4.2}$$

Following equation 4.2, the application for Costa Rica considers the following production function in which the dependent variable is the economic factor, i.e.

the revenues obtained by selling the hydro-electricity in the market:

$$lnQ = \alpha + \beta_1 \ln X_{runoff} + \beta_2 \ln X_{power} + \beta_3 \ln X_{storage} + \beta_4 \ln X_{height} + \mu \qquad (4.3)$$

where the dependent and explanatory variables are defined as follows:

Q	revenue per hectare per year for each plant (1000 x $\$_{2009}$)
Xrunoff	runoff of the watershed feeding the hydropower plant (m^3/ha·year)
Xpower	installed power of each hydroelectric plant (kW)
Xstorage	volume of the reservoir (1000 x m^3)
Xheight	downfall height, as the difference between inlet and outlet flows (m)
μ	vector of residuals
ln	natural logarithm

We ran the model using as dependent variables both hydro-energy production and annual revenues and both regressions showed similar trends in the results of the beta coefficients. Considering that private and public plants sell the energy at different prices and there are differences even within private plants, we decided to use annual revenues as dependent variable in order to estimate directly values (US$\$_{2009}$) per hectare. However, it has been our choice to use revenues as we understand it was more useful for our purposes, but it is perfectly valid to use electricity production instead, and other authors have done so (e.g. Guo *et al.* 2007).

According to Harrison *et al.* (1998), the potential hydroelectricity is directly related to runoff, therefore changes in this hydrologic attribute would directly affect the hydropower potential. However, runoff is not the only constraint, as energy production is also limited by the installed power (the capacity of the turbines) as well as by the storage capacity (Harrison and Whittington 2001). The same study suggests that climate vulnerability of hydro-energy production depends on the available storage (i.e. climate change can alter the seasonal distribution of runoff) so that, in general, the greater the storage capacity, the lower the "climate sensitivity of energy production" (Harrison and Whittington 2001: 3). We must also take into account that hydro-energy is more sensitive to reduced flows than to peak flows, that is why the storage capacity is essential. A hydropower plant has a restricted opportunity of exploiting increased flows, because its use is limited by the turbine capacity (Harrison *et al.* 2006).

In order to project potential revenues under different climate change scenarios, we make the assumption that the variables installed power, reservoir storage volume and the downfall height in Equation 4.3 remain constant for each plant, *ceteris paribus*, while runoff changes under future scenarios according to the results of the MAPSS simulations. The *ceteris paribus* assumption allows making predictions about expected impacts of one factor by screening out other disturbing factors. The runoff estimated for baseline and future projections are shown in Figure 4.2. The decrease in the runoff varies depending on the hydropower plant and the emission scenario considered.

Input data

The database was built using economic data provided by ICE (Instituto Costarricense de Energía), ARESEP (Autoridad Reguladora de Servicios Públicos), and SIEN (Sistema de información Energética Nacional de la Dirección General de Energía), enriched by the biophysical data (expressed as the mean annual runoff for the baseline year 2010) obtained from the MAPSS model (Imbach *et al.* 2010) calculated for the watershed in which each hydroelectric plant is located. The database is therefore a cross-sectional one, where only the geographical dimension is considered. This has been a limitation for the econometric analysis, not being possible to exploit the data on electricity production and revenues which were instead available in a yearly basis. Also, seasonal changes were not accounted for due to MAPSS limitations to reproduce monthly absolute values of runoff (Imbach *et al.* 2010), therefore high and low peak flows effects were not accounted for. Future research challenges could focus on using yearly or even monthly runoff data.

We start from a database with 40 hydroelectric plants in Costa Rica. However, five plants had an unexpectedly low energy generation at the reference year, which have been considered as unusual observations because the observed production does not match the installed capacity. Therefore these observations were considered to be outliers and removed from the sample, which at the end counts plants. Regarding the representativeness of these data, the selected plants represent 54 per cent of all plants and more than 70 per cent of the total hydroelectricity generated in Costa Rica in 2009 (CEPAL 2009).

Results and discussion

This section presents the results of the regression analysis, the economic value of water services used for hydro-energy, and the projected changes in hydropower production (in terms of revenues) by the end of the century for different climate scenarios. The value of water services is presented as revenues per hectare per year for each of the hydroelectric plant estimated for the baseline year 2009. Projections of revenues per hectare per year are estimated for each plant under IPCC A2, A1B, and B1 emission scenarios.

Model results

The results of the production function (equation 4.3) are presented in Table 4.1. All four variables (*Xrunoff*, *Xpower*, *Xstorage*, *Xheight*) were statistically significant and positive, as expected. This means that increased runoff, greater installed power, greater reservoirs and greater height between input and output flows translate into increased revenues per hectare per year for the hydroelectric plant.

According to the results, a 10 per cent change in runoff and installed capacity implies a 7.5 per cent and 6.8 per cent change in revenues per hectare per year, respectively. The height between input and output flows has a smaller impact: a 10 per cent change in downfall height translates into a 1.1 per cent increase in

Table 4.1 Results of the production function. Dependent variable: revenues per hectare (ln). Estimator of the model: OLS

Variable	*Coefficient*	*p-value*
$\ln X_{runoff}$	0.755**(0.081)	0.000
$\ln X_{power}$	0.692**(0.105)	0.000
$\ln X_{height}$	0.114**(0.033)	0.002
$\ln X_{storage}$	0.060*(0.034)	0.097
Constant (α)	–19.234**(1.898)	0.000

Notes: R^2 0.877; Adj. R^2 0.859; Number of obs. 33; For coefficients: ** and * indicate *t* significant at $p < 1$ and 5% respectively.

revenues flow. The same change in storage capacity turns into a 0.6 per cent increase in revenues per hectare per year.

Economic impacts under future climate change scenarios

The production function has been used to project changes in revenues associated with changes in annual runoff in the three future climate scenarios. The procedure implies using the beta coefficients showed in Table 4.1, and estimating the expected change in annual revenues per hectare consequent to the change in the runoff, as provided by the MAPSS model. Estimations of annual runoff, expected changes in the flow with respect to the baseline scenario and revenues per hectare per year for each of the hydropower plants are presented in Table 4.2. Runoff strongly decreases in all watersheds and in all three scenarios. As expected, the median reduction in runoff is smaller in the lowest emission scenario, B1 (24 per cent), and higher in scenarios A2 and A1B (31 per cent). The flow of revenues for Costa Rica in the baseline scenario is equal to 780.94 $\$_{2009}$/ha·year. This value would be reduced under the three climate change scenarios, although the lowest emission scenario -B1- shows the smallest loss (5 per cent), followed by scenario A1B (9 per cent) and the high emission scenario A2 (12 per cent). Looking at the specific plants, the results show that almost one third of the plants (11) would have increased revenues under all future climate change scenarios and 15 of them will have higher revenues under the low emission scenario B1. However, another third, twelve plants, could have losses above 30 per cent in all three scenarios.

Discussion and application at the regional scale

The relationship between forests, hydrological ecosystem services and hydropower has been assessed already by several authors who used different methodological approaches. For example, Guo *et al.* (2000) analyzed how water flow regulation and sediment retention by forest ecosystems contribute to increase electricity production in the Gezhouba hydroelectric plant (China). The services provided

Table 4.2 Annual runoff (m^3/s) and revenues ($2009/ha·year) for Costa Rica (median value) and each hydropower plant in the baseline (2009) and A2, A1B, B1 emission scenarios

	Hydropower plants	2009	A2	*Runoff (m3/s)* D(A2)	A1B	D(A1B)	m3/s	D(B1)	dfg	2009	A2	*Revenues (thousand $ (2009)·ha·yr)* D(A2)	A1B	D(A1B)	m3/s	D(B1)
1	Aguas Zarcas	2626.0	1812.9	–31.0%	1443.9	–45.0%	2131.7	–18.8%		4.81	5.34	+11.1%	4.50	–6.5%	6.03	+25.5%
2	Angostura	2157.0	1547.3	–28.3%	1518.7	–29.6%	1687.3	–21.8%		0.78	0.85	+9.0%	0.84	+7.5%	0.91	+16.4%
3	Arenal	1176.0	531.1	–54.8%	552.5	–53.0%	723.7	–38.5%		30.08	7.86	–73.9%	8.10	–73.1%	9.93	–67.0%
4	Avance	1532.0	1028.2	–32.9%	994.8	–35.1%	1308.5	–14.6%		0.22	0.09	–59.4%	0.09	–60.4%	0.11	–51.3%
5	Cacao	1854.4	1301.6	–29.8%	1312.3	–29.2%	1454.3	–21.6%		0.07	0.04	–39.7%	0.04	–39.3%	0.04	–34.4%
6	Cachi	1936.7	1315.5	–32.1%	1366.6	–29.4%	1491.4	–23.0%		0.97	0.90	–7.2%	0.92	–4.5%	0.99	2.1%
7	Caño Grande	2997.3	2203.9	–26.5%	1981.9	–33.9%	2381.0	–20.6%		0.74	1.30	+75.7%	1.20	+62.1%	1.38	+86.2%
8	Carrillos	1854.4	1301.6	–29.8%	1098.8	–40.7%	1454.3	–21.6%		0.81	0.18	–77.3%	0.16	–80.0%	0.20	–75.3%
9	Don Pedro	3127.1	2157.3	–31.0%	1828.9	–41.5%	2233.5	–28.6%		3.09	4.03	+30.3%	3.56	+15.0%	4.14	+33.8%
10	Doña Julia	2796.1	1856.4	–33.6%	1598.5	–42.8%	1989.2	–28.9%		1.55	1.44	–7.4%	1.28	–17.3%	1.51	–2.5%
11	Echandi	1759.2	1162.6	–33.9%	1026.5	–41.6%	1309.5	–25.6%		0.27	0.18	–32.9%	0.17	–38.9%	0.20	–26.6%
12	El Embalse	2664.2	1967.0	–26.2%	1962.1	–26.4%	2222.7	–16.6%		0.23	0.18	–20.4%	0.18	–20.5%	0.20	–12.7%
13	El General	2608.2	1781.6	–31.7%	1507.3	–42.2%	1935.7	–25.8%		2.10	2.70	+28.3%	2.38	+13.1%	2.87	+36.6%
14	Garita 1 1	1492.1	1060.7	–28.9%	949.9	–36.3%	1166.4	–21.8%		0.32	0.28	–12.5%	0.26	–19.5%	0.30	–6.0%
15	Garita 3 3	1216.2	747.6	–38.5%	685.6	–43.6%	831.0	–31.7%		0.75	0.49	–35.1%	0.46	–39.2%	0.53	–29.7%
16	La Esperanza	1905.1	1326.5	–30.4%	1256.6	–34.0%	1401.6	–26.4%		3.21	0.85	–73.7%	0.81	–74.7%	0.88	–72.5%
17	La Joya	1904.1	1289.2	–32.3%	1347.6	–29.2%	1471.6	–22.7%		0.23	0.35	+54.7%	0.36	+59.9%	0.39	+70.9%
18	La Rebeca	2096.0	1638.5	–21.8%	1679.3	–19.9%	1702.3	–18.8%		0.12	0.10	–18.6%	0.10	–17.0%	0.10	–16.2%
19	Los Lotes	1480.6	1013.3	–31.6%	975.3	–34.1%	1284.2	–13.3%		0.19	0.11	–40.8%	0.11	–42.5%	0.13	–29.2%
20	Miguel Dengo	1143.0	482.8	–57.8%	483.2	–57.7%	631.7	–44.7%		12.89	2.21	–82.8%	2.21	–82.8%	2.71	–79.0%
21	Peñas Blancas	1942.8	1355.1	–30.2%	1396.1	–28.1%	1407.4	–27.6%		61.88	23.40	–62.2%	23.93	–61.3%	24.08	–61.1%
22	Platanar	1870.8	1222.5	–34.7%	1190.9	–36.3%	1391.5	–25.6%		11.27	5.88	–47.9%	5.76	–48.9%	6.48	–42.5%
23	Poás	1902.6	1366.0	–28.2%	1372.6	–27.9%	1542.3	–18.9%		0.13	0.15	+17.6%	0.15	+18.1%	0.17	+28.9%
24	Pto. Escondido	1532.0	1028.2	–32.9%	994.8	–35.1%	1308.5	–14.6%		0.09	0.07	–23.4%	0.07	–25.3%	0.08	–8.1%
25	Quebrada Azul	1995.2	1363.4	–31.7%	1285.6	–35.6%	1400.6	–29.8%		0.06	0.11	+69.3%	0.10	+61.9%	0.11	+72.8%
26	Rio Lajas	2196.0	1420.1	–35.3%	1072.6	–51.2%	1598.8	–27.2%		1.63	1.60	–1.7%	1.29	–20.4%	1.75	+7.6%
27	Rio Macho	1765.2	804.1	–54.4%	1130.3	–36.0%	1333.6	–24.5%		8.48	3.47	–59.0%	4.49	–47.0%	5.09	–40.0%
28	San Lorenzo	1725.2	1149.4	–33.4%	1020.0	–40.9%	1287.7	–25.4%		0.52	0.57	+8.7%	0.52	–0.7%	0.62	+18.4%
29	Sandillal	1118.7	468.5	–58.1%	398.2	–64.4%	591.0	–47.2%		1.26	0.46	–63.7%	0.40	–67.9%	0.54	–56.7%
30	Santa Rufina	1910.7	1257.1	–34.2%	1158.1	–39.4%	1338.1	–30.0%		0.20	0.11	–45.4%	0.10	–48.7%	0.12	–42.7%
31	Suerkata	2610.2	1813.8	–30.5%	1786.2	–31.6%	1884.5	–27.8%		0.09	0.22	+134.5%	0.21	+131.8%	0.22	+141.4%
32	Toro I	2355.2	1766.7	–25.0%	1726.5	–26.7%	1967.4	–16.5%		2.28	1.03	–54.8%	1.01	–55.6%	1.12	–51.0%
33	Toro II	2513.1	1888.8	–24.8%	1479.1	–41.1%	2115.2	–15.8%		3.96	3.91	–1.5%	3.25	–18.1%	4.25	+7.3%
34	Tuis	2051.1	1361.4	–33.6%	1422.6	–30.6%	1503.8	–26.7%		0.31	0.69	+117.9%	0.71	+125.3%	0.74	+134.9%
35	Volcan	3238.9	2228.5	–31.2%	1847.7	–43.0%	2322.3	–28.3%		2.04	3.66	+79.2%	3.18	+55.5%	3.77	+84.9%
Median		**1910.7**	**1326.5**	–31%	**1312.3**	–31%	**1454.3**	–24%		**0.78**	**0.69**	–12%	**0.71**	–9%	**0.74**	–5%

generated an increase of energy production valued at 608,433 US$/year. In another study, focusing again on a hydropower plan in China (Three Gorges Hydroelectric Power Plant), Guo *et al.* (2007) estimated the benefits of flow regulation to be equal to 20.85 million US$/year in terms of increased energy production. Additionally, a saving of 14.4 million US$/year was calculated due to sediment retention by forests. The authors developed two models: the first, to estimate river water flow regulation by forestlands using several biophysical parameters; and the second, to calculate hydroelectricity generation. Our study follows a similar approach to that by Guo *et al.* (2000, 2007), in which they used a combination of biophysical and economic models. However, their study estimates changes in energy production and then multiplies this change by the current electricity price, while our production function directly estimates changes in revenues. Additionally, they focus on a single hydropower plant and do not address the potential impacts of climate change.

There are other examples analyzing the benefits provided by ecosystem services to the hydropower production. Reyes *et al.* (2001) estimated the value of hydrological services provided by forests in four watersheds in Costa Rica, in terms of runoff quantity, flow regulation and water quality for several economic activities, including hydropower. The value of all the ecosystem services addressed vary between 100 and 176 $US\$_{2001}$/ha·year based on replacement and maintenance cost valuation methods. Postel and Barton (2005) found that a hydropower company operating in Rio Volcan and Rio San Fernando watersheds in Costa Rica paid 10$/year per hectare of forest under the PES Program, although this amount represents the 25 per cent of the standard payment schemes (Pagiola 2002). Pagiola (2008) analyzed several contracts within the Payment for Ecosystem Services (PES) Program in Costa Rica, including six hydropower companies whose contribution varied from 12 to 40 $US\$_{2001}$/ha·year. Bernard *et al.* (2009) assessed the role of Tapantí Natural Park in Costa Rica in the generation of hydroelectricity in three hydropower plants (Río Macho, Cachí and Angostura). The authors estimated the monetary value that forests provided through sediment retention, amounting to 12.96 US$/ha·year in Catchí and 5.62 US$/ha·year in Angostura watersheds.[2] Finally, Locatelli *et al.* (2011) developed a methodology to map ecosystem services priorities for hydropower production in Costa Rica and Nicaragua, using fuzzy knowledge and expert knowledge to account for ecosystems service provision and demand/utility functions.

The examples shown for Costa Rica show significantly lower values per hectare than those obtained in this study as a baseline value for Costa Rica (780.94 $\$_{2009}$/ha·year). The reason is related to the methodology used. As a matter of fact, the production function is a market based approach, while Reyes *et al.* (2001) used non-market valuation methods (replacement and maintenance cost methods), which generally provide lower values than those using market base methodologies. The values obtained by Postel and Barton (2005) and Pagiola (2002, 2008) represent the payments within a PES framework, which are usually even smaller than the non-market values. The exception is the study by Bernard *et al.* (2009),

which also considers market values in terms of annual costs related to sedimentation problems. However, in our study we address a different service (surface runoff) with a different market (electricity generation), so that the two studies are not comparable in terms of results.

With regard to the impacts of climate change on hydropower, several studies have addressed this issue using different perspectives (local or regional). A first group of studies focus on specific/individual hydropower plants or river basins and are therefore developed on a local scale. For example, Muñoz and Sailor (1998) developed a statistical model to link climate variables and hydroelectric generation in three watersheds in northern California. Based on historical data, the model was then used to project electricity production under future climatic scenarios. For a 2°C increase in temperature and 20 per cent reduction of precipitation, they estimated significant impacts on hydropower availability, up to 50 per cent in the winter and 5 per cent in spring. Harrison and Whittington (2002) assessed the technical and financial viability of several hydropower projects in the Zambezi River (shared by eight countries in Africa) under different climate change scenarios. According to their estimations, mean monthly sales of electricity by 2080 would decrease between 6 per cent and 22 per cent, depending on the global circulation model used. Schaefli *et al.* (2007) addressed the impacts of climate change on a water resource system and its management at the scale of a single hydropower plant located in the Swiss Alps. In their approach they use four models: a water management model, a hydrological model, a glacier surface evolution model and a model to develop a local scale meteorological time series. For the period 2070–2099 they estimated a 36 per cent decrease of hydropower production compared to the control period (1961–1990). Vicuna *et al.* (2008) developed four hydrologic scenarios to estimate the impacts of climate change on a hydropower station in California. Their results show, as expected, a smaller energy production (and revenues) in drier climate change scenarios of about 10 per cent compared to historical values, while revenues increase in a similar amount in the wet scenario. They also found out that storage plays a key role to in order to soften the effect of low stream flow. Maurer *et al.* (2009) estimated the change in temperature and precipitation in the Rio Lempa basin (Guatemala and Honduras) for B1 and A2 emission scenarios. By using a land surface model the authors translated these changes in temperature and precipitation into hydrologic impacts, specifically looking at variations on the inflow to two major hydropower reservoirs by the end of the century. According to their results, reservoir inflows would be reduced by 13 per cent (B1) and 24 per cent (A2) by 2070–2099. From an adaptive perspective, Minville *et al.* (2009) analyzed the changes in the hydrological regime of the Peribonka River (Canada) as a result of climate change including in the scope of their study the operating rules of reservoir management. Four types of models were used in this study: a water management model, a hydrological model, a glacier surface evolution model and a model to develop a local scale meteorological time series. Their results present a declining trend of 1.8 per cent in the short term (2010–2039) but annual and seasonal increasing trends in the mid-

(9.3 per cent for the period 2040–2069) and long-terms (18.3 per cent between 2070 and 2099).

A second group of studies provide a regional assessment using different methodological approaches. This is the case of Lehner *et al.* (2005), who estimated a 6 per cent reduction of Europe's gross hydropower potential by 2070 using the WaterGAP global integrated water model. This model produces time series of river flows based on climate change and socioeconomic scenarios. De Lucena *et al.* (2009) assessed the impacts of climate change on the Brazilian hydropower sector based on historical flow data using a selection of reference hydroelectric plants and future rainfall projections. A small fall in energy generation (less than 2.5 per cent) was obtained with an operation simulation model (SUSHI-O) production within 20 plants of the river basin. Estimated annual losses by 2080 vary between 2.57 per cent (scenario B1) and 3.15 per cent (scenario A1B).

Some of the studies analyzed above use a similar approach to that presented in this chapter, such as Guo *et al.* (2000, 2007), but the scope of their analysis is usually limited to a single or a few hydropower plant. Additionally, they do not assess the impacts of climate change. Other authors consider the changes that global warming may cause, but then use different methodological approaches none of which addressed the impacts of climate change on the hydroelectric sector by using a combination of biophysical and economic models. The approach proposed in this chapter uses a simplified hydrological approach since MAPSS model does not account for runoff routing nor aquifer recharge and discharge which could play an important role in runoff estimation for some catchments. Although the MAPSS model calibration was performed using runoff data from a set of catchments that aimed at representing the climate gradient of the Mesoamerican region (Imbach *et al.* 2010), and therefore allows to cover several catchments, its specific performance over Costa Rica was not assessed. Another source of potential errors in runoff estimates was found over smaller catchments, possibly due to larger relative effects of catchment delineation errors (Imbach *et al.* 2010). This sums up to the uncertainty of climate change projections and that associated with the economic model, which considers the current energy market situation. Technological changes and future market trends are not considered.

The methodology presented here can be very useful for future trend analysis at the regional scale and for raising awareness about potential impacts of climate change. Presumably, the most difficult issue for upscaling this approach to Central America is related to the gathering of economic information related to the energy produced by each station and the revenues generated. The Economic Commission for Latin America (CEPAL), one of the five regional commissions of the UN in the region, publishes the electricity statistics of all Central American countries in a yearly report. This report should be a good starting point for collecting some of the missing information. Lastly, some policy implications of the results are discussed in Box 4.1.

Box 4.1 Policy implications

The main contribution of this study is that it provides a reference of the magnitude of expected impacts of the Costa Rican hydropower sector to climate change, while identifying as well the hydroelectric plants with the highest vulnerability. The results show a very significant reduction of the hydrological regime, and thus, a general reduction of hydropower production in Costa Rica. Considering that the hydropower sector in Costa Rica generated in 2013 the 68 per cent of the energy produced in the country, defining adaptation policies and measures to address these impacts is unavoidable.

From a *strategic* perspective, it seems reasonable to suggest that Costa Rican authorities should consider other energy alternatives with lower vulnerability that would gradually be complementary to hydropower. Technology could also play an important role, especially focused on increasing the productivity of existing hydro-energy stations. Thus, the reduction of runoff could be compensated with a greater operating efficiency. Our study can be of great interest to policy-makers as it can help identifying the hydropower plants that may be more vulnerable to climate change in Costa Rica.

In relation to the *design* of new infrastructures, new proposals should necessarily consider the effects of climate change and analyze their feasibility from an economic and environmental perspective under the new conditions (Lehner *et al.* 2005). Infrastructures that have reservoirs are less vulnerable, but its greater environmental impact should also be accounted for. Decisions should be taken from a comprehensive and integrated perspective that contemplates water, environment, energy and their interrelationships (Haas 2009). Additionally, a significant reduction in the hydrological regime may turn water into a critical resource, not only for the energy sector, but also for agriculture or human consumption. Hence, future designs should also address competing environmental services and beneficiaries.

From an *operational* standpoint, it is possible to modify the operation rules in order to adapt to a change in water availability, both in relation to the water volume as well as to the timing and seasonal changes of runoff. The management of the reservoirs can be changed to optimize the response to low flows. Also, mean levels of the reservoirs might be changed in each season to maximize the production: levels can be lower when there is more resource, avoiding unproductive spills, and higher when input flows decrease (Minville *et al.* 2009).

Finally, further *research* should be carried out to improve biophysical and economic models, as well as data collection, at the regional or national scale, as an input for policy and strategic decision-making, but also at the local scale, in order to improve the information about impacts on specific watersheds or hydropower stations.

Conclusions

This chapter presents a methodological approach to assess physical and economic impacts related to changes in water supply for hydropower due to climate change. A theoretical model is developed and discussed in light of the existing literature, and is then applied to an empirical context examining hydroenergy in Costa Rica.

A sample of 35 plants has been constructed for this purpose. The methodology combines a bio-physical model (MAPSS) estimating the expected runoff changes in future climate scenarios with an economic one based on the production function which relates the quantity of water available (runoff) with the energy generated by the selected plants. Changes in the energy production have been then modelled and projected using the production function under different future IPCC scenarios.

The model developed allows to assess the economic impacts of climate change on the hydroelectric sector, using the association between bio-physical data, technical data related to the plants (installed power, downfall height, volume of the reservoir) and economic inputs (in terms of revenues produced by each plant).

Results show a significant reduction in runoff in all future scenarios that varies between 24 per cent (B1) and 31 per cent (A1B and A2). This translates in a reduction of the expected revenues of the hydroelectric sector in Costa Rica under all climate change scenarios (A2, A1B, B1). The reduction is however lower in the B1 scenario (5 per cent), which incorporates sustainability criteria, while the impact is greater on the scenarios of group A which are more economically driven, with losses reaching 9 and 12 per cent under scenarios A1B and A2 respectively. It is important to notice that the expected impacts can be quite different from plant to plant, depending on the projected decrease of runoff in the respective watershed and the technical characteristics of each plant. In fact, almost one third of the plants would increase their revenues under all three emission scenarios.

The model offers a tool that can be easily adapted to other geographical contexts, or to assess the impacts on specific hydrologic plants in Costa Rica. One of the strengths of this study is the close relationship between biophysical data/modelling, and economic analysis. However, we must be aware of the uncertainty linked to climate and ecological models, and that this is also affecting economic modelling. Cumulative uncertainty should also be considered, as each step (input data, climate models, economic impacts…) may add uncertainty to the previous one (Refsgaard *et al.* 2013).

It has been necessary to limit the scope of this study as future projections were estimated based solely on changes in the amount of runoff, regardless the evolution of the energy market in the long term. Adaptation options through changes in the operation and regulation of reservoirs, as analyzed, for example, by Minville *et al.* (2009) in the Peribonka River watershed, were not considered as all the data needed for such a detailed analysis were not available for our national scale study.

Nevertheless, changes in discharge appears to be a suitable indicator to assess whether the impacts of climate change will contribute to a "general growth or decline in the hydropower potential" (Lehner *et al.* 2005).

Another limitation is that in the construction of the database we could not incorporate annual and even monthly production data (runoff and electricity). However, based on the reviewed literature, we believe the four variables used (*runoff*, *power*, *storage* and *downfall height*) are the key variables to be considered. Further research should also incorporate data on an annual basis (even monthly, to be able to catch seasonal trends) in order to improve and calibrate the production function used.

Finally, we should also notice that the estimated economic impacts are due solely to the effect of a change in the provision of water, all other things being held constant. As a matter of fact, the model does not incorporate effects such as changes in energy prices, the country's economic growth, and others. Additional changes in other socioeconomic variables could both mitigate or exacerbate the effects of climate change.

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Notes

1 Each of the three scenarios considered in this study (A2, A1B and B1) represent different future development pathways as described by the IPCC on its Special Report on Emission Scenarios (IPCC 2000). A2 and A1B are economically oriented scenarios, focused on material consumption. The first represents a very "heterogeneous world" with a "regionally oriented" economic growth, while the latter is more "globally

oriented" and considers a balance between the different energy sources. The "environmentally oriented" scenario B1 is less "material intensive," focusing on sustainability and equity.

2 These values have been estimated dividing the avoided costs provided in the reference study by Bernard *et al.* (2009) by the total hectares of the watersheds considered.

5 Climate change economic impacts on water and recreation services in Central American forests

Elena Ojea, Juan Carlos Zamora, Julia Martin-Ortega and Pablo Imbach

Introduction

The estimation of the economic value of ecosystem services is expected to play an increasingly important role in conservation planning and ecosystem-based management (Plummer 2009; Stenger *et al.* 2009) in order to ensure that human actions do not damage the ecological processes necessary to support the continued flow of ecosystem services on which welfare of present and future generations depends (MEA 2005; Daily and Matson 2008). This becomes particularly relevant under the threat of climate change, where a 3°C warming is estimated to transform about one-fifth of the world's ecosystems (Fischlin *et al.* 2007).

Economic valuation of ecosystem services requires up-to-date and reliable information and considerably better understanding of the landscapes that provide such services (Troy and Wilson 2006, in Baral *et al.* 2009). In this line, several studies have given attention to the landscape functions in order to calculate benefits associated to ecosystem services. Based on these needs, the focus of economics has so far been placed on single resources with commercial use (land, fisheries, forests, energy, etc.) and goods and services provided by nature in the absence of markets (clean air, aesthetics, recreation). However, recent work is shifting the stand of economic analyses, and attention is now being paid to understand the biophysical underpinning of ecosystem functioning and how land use affects this, to predict the provision of services and their value (Polasky and Segerson 2009; Naidoo *et al.* 2009; Naidoo and Iwamura 2007). Therefore, complex ecological functions and processes have started to be considered from an economic perspective where traditionally that ecological level wasn't part of an economic analysis. This awareness lead to the recent interest in integrating ecological and economic sciences (Polasky and Segerson 2009), but both the quantification of the service provision and the values of these services has proven difficult (Nelson *et al.* 2009). These developments combine the use of spatial data such as vegetation types, land use, productivity, etc. with economic valuation in order to provide more accurate estimates of ecosystem services (Naidoo *et al.* 2009; Egoh *et al.* 2008; Nelson *et al.* 2009). Despite the previous work on spatially based ecosystem service valuation (e.g. Brouwer *et al.* 2010; Martin-Ortega *et al.* 2012; Nelson *et al.* 2009; Bateman *et al.* 2013), there is a lack of studies looking at how the economic value of

ecosystem services is expected to change under expected climate change (Ding *et al*. 2010).

Some methodologies of environmental valuation allow researchers to understand the benefits of ecosystem services that are not traded in existing economic markets. This is needed to better understand the role of ecosystem services and help policy-makers to plan for the sustainability of natural resources. A lack of economic valuation could underestimate the importance of such resources and leave to a detriment on the ecosystem services supply. Non-market values are associated with many ecosystem services such as water quality, recreation, flow regulation, conservation of wild species, and they amount to an important share of the benefits from ecosystem services (TEEB 2010). However, only certain methodologies can provide information on their economic value as they are usually not traded in existing markets, as in the case of recreation or carbon sequestration (Zandersen and Tol 2009; TEEB 2010). This chapter focuses on recreation and water services provided by forest ecosystems as these entail important economic values that are many times omitted from policy decision-making.

The aim of this chapter is to estimate the potential economic impacts of climate change on water and recreation services in tropical forests of Central America. We conduct this analysis by looking at how forests provide economic benefits through water and recreation services and, based on their dependence on forest types and area, we estimate changes in their economic value under different climate scenarios altering those forest types, and therefore, the change in economic benefits derived from them. We do this by collecting primary valuation studies on the region and conducting a meta-analysis where economic data is enriched with geographic information and a vegetation model. By means of benefit transfer techniques, we project economic values under different scales and scenarios of climate change.

The chapter begins by introducing the methodology used, including details on the database construction, the addition of biophysical information, the econometric analysis, the scaling-up and the ecosystem benefits estimation under the projected scenarios of climate change. We then present the final results with the changes in ecosystem services benefits due to climate change, before summarizing the conclusions and presenting recommendations for future research.

Methods and analysis

The analysis conducted here follows six main steps that are illustrated in Figure 5.1. A database is built from original primary published studies and reports valuing water and recreation services in tropical forests. The economic data are combined with spatial and vegetation information (Holdridge zones)[1] to identify areas and forest types for each observation in the database. This database is explored and analyzed conducting an econometric analysis with a meta-regression. Results from the meta-analysis are used for scaling up and benefit transfer of the economic benefits from water and recreation services. While scaling up here refers to the transference of economic values from the study site to a broader area, benefit

transfer refers to the exercise of estimating the economic values under different conditions due to climate change affecting vegetation types. Climate change projections of the vegetation types are used to estimate future changes in the values of water and recreation services. Finally, we compare the results for the baseline and climate change scenarios in order to estimate the economic impacts on these services. Each step is described in detail below.

Database construction

A database on values for tropical forest ecosystem services was constructed from the existing available literature citing case studies from Central and South America. These studies were systematically compiled and reviewed for allowing the meta-analysis. Ecosystem services were identified and classified accordingly. The following sections detail this process.

Literature review

A review on the literature on valuation of tropical forests was developed as a first step. Three main criteria were followed in order to include the studies in the database:

- the source of the study;
- matching keywords; and
- identifiable ecosystem service and economic value.

THE SOURCE OF THE STUDY

Five main sources of studies were consulted and observations were only included if coming from one of them. These sources are the following:

- *EconLit*, one of the broader databases on economic literature, including more than 480 full-text journals, with many that are unavailable from other full-text databases (http://search.ebscohost.com).
- *ScienceDirect*, full-text scientific database offering journal articles and book chapters (www.sciencedirect.com).
- *EVRI*, a database with valuation studies worldwide (www.evri.ca).

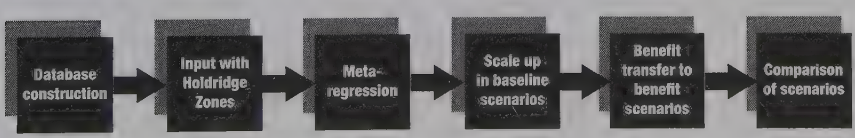

Figure 5.1 Methodological approach

- *FAO Forest Studies*, where FAO collects information about the value of the environmental benefits of forests (http://foris.fao.org/valuation/search/index.jsf).
- *Clase and Periodica*, which contains documents published in Latin American journals specialized in social sciences and humanities, and science and technology, respectively (http://dgb.unam.mx).

MATCHING KEYWORDS

In every source of original studies, we conducted a systematic search by entering selected keywords. These keywords were directed at identifying the ecosystem type, the ecosystem services of interest, and the set of methodologies that can be applied for the valuation of ecosystem services. Table 5.1 shows the keywords employed in the systematic search.

IDENTIFIABLE ECOSYSTEM SERVICE AND ECONOMIC VALUES

Studies were selected when the service being valued was specified or/and could be re-classified using the available information on the source. In the same line, we looked for the units of value such that only papers where we can clearly identify the periodicity of the benefits and their units of measurement (per household, or per hectare, etc.) were selected. These criteria may look straightforward at first but some studies did not provide enough information in this respect, and therefore were not considered for the analysis.

From this process, a total number of 43 original studies valuing non-market services from tropical forests were selected and included in the database. Geographically, we selected studies in South and Central America, as we were interested in the ecological consistency of the sites. We collected information on

Table 5.1 Keywords employed in the review search

General	*Services*	*Goods*	*Method*
Forest	Provisioning	Water	Economic valuation
Forest ecosystems	Cultural services	Soil	Contingent valuation
Tropical forest	Recreation	Biodiversity	Choice experiments
Benefits	Non-use values	Bioprospecting	Market
Services	Regulating		Travel cost
Ecosystem services	Flood control		Hedonic prices
Woodland	Watershed control		PES
Protected areas	Erosion control		Stated preferences
Latin America	Supporting		Revealed preferences
Country names*	Conservation		

Note: *Panama, El Salvador, Guatemala, Nicaragua, Belize, Honduras and Costa Rica.

many variables; including the value per hectare per year, the study site and its area, or the valuation method employed. The studies include a range of valuation techniques from cost-based methods (including avoided costs, reforestation costs and opportunity costs) to non-market stated preference techniques (mostly contingent valuation) and market data (e.g. hydropower and agricultural production). We have also included in our dataset a number of studies on payments for ecosystem services (PES) as proxies of market-values.[2] From these 43 primary valuation studies we obtain 132 value observations. This number is substantial given the geographical restriction of our dataset. Most studies are published and have been conducted over the last 20 years. Table 5.2 shows the variables for analysis that have been selected, together with the description and coding and the format in which this information has been introduced in the database. Additional variables were added to the database from international statistics at a country level, such as income and population density.

Targeted services

For this exercise we focus on forest ecosystem services that are not directly valued in conventional markets, and therefore can only be approached from the non-market economic literature. These include services that can be measured with methods of stated and revealed preferences, as well as created environmental markets such as PES or bio-prospecting. Although information about all these services is collected, the exercise in this chapter focuses on water and recreation values from tropical forests. This is due to two constrains: the availability of water and recreation benefits in forests is larger than for the other services and allows us to conduct a quantitative analysis; and both water and recreation services share a common economic rationale, where both services provide use values for the people that benefits.

We classify our observations from original studies into final ecosystem services using the output based classification. A detailed description of how this has been done is available in Ojea *et al.* (2012). The reclassification dropped the number of studies from 43 to 39 but has the advantage of avoiding double counting problems among studies and services (see Box 5.1). What follows is a detailed description of the types of studies and services we are looking at.

Water services

Forests provide crucial services related to the maintenance of the hydrological cycle and the services provided by watersheds. In general terms water services can be defined as the benefits to people produced by terrestrial ecosystems effects on freshwater (Brauman *et al.* 2007). The progressive loss of these services risk harm to human health through lowered drinking water quality, higher water costs that may burden poorer populations and lower crop productivity and hydroelectric output from reduced dry-season flows. These services are particularly important for communities in the tropics, either because rainfall is highly seasonal or locally

Table 5.2 Description of the variables included in the database

	Item	*Description*	*Format*
Source	Full reference of paper – authors, year, title, reference	Full reference	Text
	Authors	e.g. 'F. Bèrnard, R. S. de Groot, J. J. Campos'	Text
	Year of publication	e.g. 2009	Text
	Published	If the study is published on a journal (1); rest (0)	Dummy
Good	Ecosystem service (covered by the study)	The specific ecosystem good and service valued (water regulation, carbon sequestration, etc.)	Text
	MEA classification	Provisioning, regulating, cultural or supporting (if possible), and combinations (when values refer to more than one service)	Text
Scenario	Environmental change	Description of the change: if it is a gain, avoid losses, a qualitative change or a quantitative change. Description of the scenario: the baseline level, the target level, the units of measure.	Text
	Value for	Residents, visitors, stakeholders, owners, society	Text
	Threat	Environmental pressure affecting the EGS: climate change, deforestation, land use fragmentation, fires.	Text
Forest characteristics	Land use type	If forest is 'managed' or 'natural'	Text
	Forest size	Number of hectares of the forest being valued	Number
	Forest status national	If forest is protected, what status (National)	Text
	Forest status international	If forest is protected, what status (International)	Text
	Biodiversity hotspots	If forest is a biodiversity hotspot	Dummy
Monetary value	Value of the EGS	Monetary value of the ecosystem service	Number
	Units used in the study	e.g. US$/ha	Text
	Range of values	Monetary interval	Text
	Time basis of value	Annual value vs. total value	Text
	Year of value	e.g. 2006	Number
	Standardized value – units used	ex. 2000 US$/ha	
	Standardized value	Standardized value (converted to the standardized units)	Text
	Economic value	Use value (direct use, indirect use, option), non-use value (existence, bequest, altruistic), combinations. TEV	
Method	Valuation method	Detailed valuation method (CV, CE, travel cost, etc.)	Text
Location	Region	Specific region of the study (valuation)	Text
	Country of study	Country	Text

Box 5.1 Water services classification for the purposes of the economic valuation

Water services are considered here as the final service providing a benefit that is given a value, following the output based classification literature which is a built upon the Millennium Ecosystem Assessment classification (Brauman *et al.* 2007; Ojea *et al.* 2012). Under this classification, water services can be under four groups:

- *Improvement of extractive water supply:* water supply is a provisioning service describing ecosystems modification of water used for extractive and *in situ* purposes, which include municipal, agricultural, commercial, industrial and thermoelectric power use.
- *Improvement of in-stream water supply:* In-stream water supply includes hydropower generation, water recreation and transportation, and freshwater fish production.
- *Water damage mitigation:* Water damage mitigation is a regulating service; it includes ecosystem mitigation of flood damage and of sedimentation of water bodies, saltwater intrusion into groundwater and of dry-land salinization.
- *Provision of water-related cultural services:* Cultural hydrologic services include spiritual uses, aesthetic appreciation and tourism.

limited, or because intensively cultivated and densely populated agrarian landscapes downstream are affected by soil-hydrological process in the upstream forest (Bonell and Bruijnzeel 2004). See Table 5.3 for a complete list of water services studies and their classification.

Cultural and provisioning services

Cultural and provisioning services are of key importance both for the economic returns associated with the direct enjoyment of forests, but also for conservation purposes and market applications like bio-prospecting. Tropical forests sustain significant recreation values that are many times defended as a way to promote conservation against other depletive uses (Gössling 1999; Hearne and Salinas 2002). Indeed, protected areas have been proposed as one of the main policies to halve deforestation (Adams *et al.* 2008). Nations promote nature-based tourism with the aim of increasing income in line with the objective of nature conservation (Hearne and Salinas 2002). Nature tourism is recognized as an important ecosystem service (MEA 2005) generating resources for conservation and local development (Goodwin 1996; Gössling 1999). Biodiversity prospecting is both a mechanism for discovering new pharmaceutical products and saving endangered

Table 5.3 Studies valuing water ecosystem services

Reference	*Ecosystem Service*	*Country*
Adger *et al.* (1995)	Water damage mitigation	Mexico
Asquith *et al.* (2008)	Improvement of extractive water supply; in stream water supply and supporting	Bolivia
Barrantes and Castro (1998a)	In stream water supply; improvement of extractive water supply	Costa Rica
Barrantes and Castro (1998b)	All water services	Costa Rica
Barrantes *et al.* (2003)	All water services	Costa Rica
Barrantes and Castro (1999)	Improvement of extractive water supply	Costa Rica
Chomitz *et al.* (2005)	In stream water supply	Costa Rica
Corbera *et al.* (2007)	In stream water supply and water damage mitigation	Guatemala
De Sena (1997)	Water related cultural services	Costa Rica
Johnson and Baltodano (2004)	Improvement of extractive water supply	Nicaragua
Kosoy *et al.* (2007)	Improvement of extractive water supply	Honduras; Costa Rica; Nicaragua
Marozzi (1998)	Improvement of extractive water supply	Costa Rica
Martínez *et al.* (2009)	In stream water supply	Mexico
Mejías *et al.* (2000)	Improvement of extractive water supply; in stream water supply	Costa Rica
Merayo (1999)	Improvement of extractive water supply	Costa Rica
Moreno (2006)	All water services	Costa Rica
Pagiola (2008)	In stream water supply; improvement of extractive water supply; improvement of extractive water supply	Costa Rica
Postle *et al.* (2005)	Improvement of extractive water supply; in stream water supply	Ecuador; Costa Rica
Reyes *et al.* (2001)	Improvement of extractive water supply and in stream water supply	Costa Rica
Reyes *et al.* (2004)	All water services	Costa Rica
Reyes and Cordoba (2000)	In stream water supply	Costa rica
Solórzano *et al.* (1995)	Improvement of extractive water supply	Cosa Rica
Valera (1998)	Improvement of extractive water supply	Costa Rica
Vargas (2004)	Improvement of extractive water supply; in stream water supply and supporting	Bolivia
Veloz *et al.* (1985)	Water damage mitigation	Dominican Republic
Whittington *et al.* (1990)	Improvement of extractive water supply	Haiti

Source: Ojea *et al.* (2012).

ecosystems (Simpson *et al.* 1996). Despite its importance, data on the value of these transactions is difficult to obtain and studies valuing the benefits of this service are very relevant for the analysis. Box 5.2 describes the specific services that we collect from the literature, and Table 5.4 presents the studies considered under those services with the reference, the final classification of services and the country of the study.

Box 5.2 Cultural and provisioning services classification for the purposes of the economic valuation

For the purposes of our analysis, we identify the following ecosystem services as obtained from the literature review:

- *Recreation and ecotourism:* Tropical forests provide large recreation values derived from local and foreign visitors to protected areas. The main problem on quantifying these values arises since recreation is dependent on the specific forest area, where not all forests are suitable for recreation and most of times recreation only takes place in protected areas. Here we identify the three different specific services that may arise:
- *Recreation in protected areas:* many studies value recreation values from protected areas using non market valuation methods such as travel cost or contingent valuation. Also, in many protected areas in Central America the policy is of asking for an entrance fee, which directly provides an estimation of the recreational value of those areas.
- *Ecotourism:* ecotourism refers to sustainable tourism in natural areas.
- *Aesthetic:* aesthetic values are related to the scenic beauty of tropical forests and values are derived from the direct enjoyment of the scenery.
- *Conservation values:* conservation values refer to the economic benefits of conserving wild species and habitats, including existence values.
- *Bioprospecting:* Biodiversity prospecting or Bioprospecting can be defined as the search for chemicals produced by wild organisms. These components have a high potential for commercial use in the industrial and pharmaceutical applications.

Biophysical information

The main aim of this step is to introduce a biophysical variable which is expected to have a significant effect on the ecosystem service flow and therefore on the economic value of the studied forest ecosystem services. The value of ecosystem services is expected to vary and depend on the characteristics of the forest, including its extension and forest type. This has been seen in many previous meta-analysis in forests (Barrio and Loureiro 2010; Ojea *et al.* 2010), but not only

Table 5.4 Studies valuing recreation and conservation ecosystem services

Authors	*Ecosystem service*	*Country*
Bernard *et al.* (2009)	Recreation, bioprospecting	Costa Rica
Chase *et al.* (1997)	Recreation	Costa Rica
Echeverria *et al.* (1995)	Recreation	Costa Rica
Shultz *et al.* (1998)	Recreation	Costa Rica
Menkhaus and Lober (1996)	Ecotourism	Costa Rica
Tobias and Mendelsohn (1991)	Ecotourism	Costa Rica
Adger *et al.* (1995)	Recreation; conservation	Mexico
Ortiz *et al.* (2001)	Recreation	Brazil
Holmes *et al.* (1998)	Recreation	Brazil
Adams *et al.* (2008)	Conservation	Brazil
Hearne and Salinas (2002)	Scenic beauty; recreation	Costa Rica
Bienabe and Hearne (2006)	Biodiversity protection; scenic beauty	Costa Rica
Horton *et al.* (2003)	Conservation biodiversity	Brazil
Simpson *et al.* (1996)	Bioprospecting	Ecuador, Brazil, Colombia, Chile
Carranza *et al.* (1996)	Conservation	Costa Rica
Adamson (2001)	Recreation; enjoyment	Costa Rica

(Brander *et al.* 2007). However, this link has been frequently done based on an area proxy, as very few meta-analyses have been spatially explicit (incorporating the exact distribution and magnitude of variables in space), and employing proxies such as the total size of the site or the protected areas in the country of study. In this exercise we overcome these problems by including specific spatial data from a biophysical model, in the form of Holdridge zones (Tropical Scientific Center 2005). This supposes a new methodological improvement that will allow us to better understand how the benefits from ecosystem services can be explained by key factors related to the provision of these services.

Holdridge zones are potential vegetation types determined with climatic variables (Holdridge 1947, 1967), specifically, mean annual biotemperature (temperature range allowing vegetative growth), annual precipitation and humidity provinces (potential evapotranspiration ratio to mean annual precipitation) (see Box 5.3 for an introduction to the Holdridge zones system). Holdridge life-zone system has been used to address potential impacts of climate change on vegetation distribution (Yates *et al.* 2000), to parameterize vegetation units of a continental water balance model (Yates 1997) and estimate economic impacts of climate change on protected areas (Velarde *et al.* 2005). Holdridge zones resulted a

practical biophysical information for the purposes of the current analysis, although other possibilities were discussed.

Box 5.3 The Holdridge life zones system

The Holdridge life zones define potential vegetation types (climax vegetation) according to climate long-term averages based on a series of hexagons formed by the intersection of intervals within a logarithmic axis framed in a triangular coordinate system. The classification of vegetation is determined by the mean annual bio-temperature and precipitation. The third variable is given by a humidity gradient determined by a precipitation–temperature relationship (Holdridge 1967; Holdridge *et al.* 1971). The biotemperature estimates the average temperature where vegetation activity occurs and therefore is the mean temperature within the 0–30°C range.

Changes in vegetation types (due to climate change) based on Holdridge life zone system was made following these steps using data from the high-resolution WorldClim climatology (Hijmans *et al.* 2005):

- Altitudinal belts were estimated based on mean annual biotemperature.
- Humidity provinces were estimated based on total annual precipitation.
- Latitudinal region were estimated based on potential evapotranspiration ratio, and was calculated by estimating biotemperature at sea level.
- Life zones map: final maps were built by combining the three previous variables using current (baseline) and future climate projections (future). Maps were coded and the area for each life zone was estimated per study site for further analysis.

In order to simplify the economic analysis, Holdridge zones were grouped in four main types. These types vary according to their characteristics in: dry, moist, wet and rain forests. A more detailed classification of Holdridge zones was not possible due to the number of observations and also the lack of sufficient information from the original studies which studied only a section of a site that can correspond to different forest types. Also, Holdridge zones information was not available in all studies since observations from Southern latitudes did not match the objectives and from the 132 value observations, 84 observations are employed in the subsequent analysis. This reduction is very significant but necessary in order to have accuracy in the vegetation types and allows us to conduct more reliable benefit transfers. Table 5.5 depicts the life zones grouping into four categories that we are using for the economic analysis.

Table 5.5 Equivalence and classification of Holdridge Zones

Holdridge zone	*Variable*
Tropical very dry forest	Dry forests
Premontane dry forest	
Lower montane dry forest subtropical	
Lower montane dry forest	
Tropical dry forest	
Premontane moist forest	Moist forests
Lower montane moist forest subtropical	
Lower montane moist forest	
Tropical moist forest	
Montane wet forests	Wet forests
Premontane wet forests	
Lower montane wet forests subtropical	
Lower montane wet forests	
Tropical wet forests	
Montane rain forest	Rain forests
Premontane rain forest	
Lower montane rain forest subtropical	

Biophysical data and value observations in our dataset were combined by means of geographical information systems. We geo-referred each observation from the database described below, where each observation from a site was given that site's location. Once sites were mapped, by means of spatial analysis we identified the potential Holdridge zones present by site, as well as their area.

Information on the Holdridge zones was however not available for all the observations on the dataset and this has significantly reduced our sample. See Box 5.4 for a discussion on the trade-offs among feasibility and accuracy.

Meta-analysis

Meta-analyses are a widely applied analytical tool to summarize the findings of the literature for a specific field. In environmental valuation, meta-analyses have been applied to study the variables that are affecting values in a range of ecosystems such as wetlands (Woodward and Wui 2001), coral reefs (Brander *et al*. 2007), or for biodiversity (Ojea and Loureiro 2011; Loomis and White 1996). Several examples have been developed for forest ecosystems, such as looking at the effect of conservation programs over values (Barrio and Loureiro 2010), the effect of biodiversity (Ojea *et al*. 2010), looking at regional characteristics of the forests that explain their benefits (Lindhjem 2007), or looking at one specific service such as the recreation values from forests (Bateman and Jones 2003; Zandersen and Tol 2009). However, no studies have looked at the ecosystem services values from tropical forests in Central America, by including both market and non-market

Box 5.4 Trade-offs in feasibility and accuracy of economic estimations when including biophysical information

Although it is important to include biophysical information in order to improve the accuracy of economic analysis, this information is many times difficult to access and process and many times even unavailable. It is important to identify the most accurate information that is also available, meaning there is a trade-off on the improvement of a methodology and the feasibility of obtaining results in time. An illustration of this trade-off would be questioning what biophysical variables we can project under future climate scenarios. In the case of the present exercise, water services are related to forest area but also to another set of different variables such as runoff. Forest area and forest type are related to the provision of water services since different forest types are related to different levels of ecosystem services and space is a proxy for provision quantity. Water runoff however is a different biophysical unit that is directly linked to the available water in the forest and the provision of water supply. An alternative to Holdridge zones would be including runoff projections under climate change, which proves to be more complex (Imbach *et al.* 2012) and does not represent all the services of this study. Further applications of this methodology can explore whether different variables lead to different economic results.

methodologies on a regional geographical scale. The present study bridges that gap by combining the economic literature with the paired biophysical information in order to understand the characteristics under which forests provide different benefits for ecosystem services.

Meta-regression function

From the database constructed in the previous sections, and based on the lessons learned from the design and the results of previous environmental valuation studies (de Groot *et al.* 2012; Brander *et al.* 2006; Lindhjem 2007; Ojea *et al.* 2010; Ojea and Loureiro 2011), we analyze our data in order to explain what variables are affecting water and recreation values. We follow the recommendations on good practices in Meta-analyses from Nelson and Kennedy (2009), and estimate the following function:

$$\ln Y = \ + {}_{sa}X_{sa} + {}_{zv}X_{zv} + {}_{c}X_{c} + u \qquad (5.1)$$

where Y is the vector for the values of the ecosystem services, and the X terms are the explanatory variables which can be summarized in three groups: X_{sa} for the characteristics for the environmental services, X_{zv} for the characteristics of the

Holdridge zones and X_c for the characteristics of the context or country where the valuation took place. The variables included in the meta-regression are summarized in Table 5.6.

Table 5.6 Variables included in the meta-analysis

Source	*Variable*	*Description*	*n*	*Mean*	*St. dev.*
Valuation studies	*lnvalueha*	Logarithm of the benefits per hectare per year	111	3.4331	2.5454
	water	Water service (1) rest (0)	131	0.6489	0.4033
	recreation	Recreation services (1), rest (0)	131	0.1679	0.3752
	*otherservice***	Conservation services (1); rest(0)	131	0.1832	0.3883
	lnareasite	Study site area in logarithm form	103	10.1408	2.3031
Holdridge Zones	*lnareahz*	Area of the main Holdridge Zone (HZ) in the study site (in logarithm)	84	9.3579	1.2805
	maindry	Main HZ is dry (1), rest (0)	84	0.1466	0.3561
	mainmoist	Main HZ is moist(1), rest (0)	84	0.2267	0.4215
	mainrain	Main HZ is rain (1), rest (0)	84	0.0667	0.2511
	*mainwet***	Main HZ is wet(1); rest (0)	84	0.5600	0.4997
WB*	*lnincome*	Per capita income in the country of study in logarithm form	108	8.3631	1.0700
	lnpop	Population density in the country of study in logarithm form	108	-0.5083	0.6588

* World Bank data: IMF (2011), country-based data.
** Omitted variables on the model.

Meta-regression results

We use a generalized least square (GLS) model treating the data as panel data, following Nelson and Kennedy (2009). The estimated coefficients are summarized in Table 5.7. We restrict the analysis to the variables of interest for the benefit transfer exercise, omitting the methodological variables. We have favored the inclusion of the characteristics of the Holdridge zone with a variable indicating the size and three dummies for the Holdridge zone type. The Holdridge zone considered for each case study is the larger one present at the study site, so this is the predominant or main Holdridge zone. Estimated coefficients carry the expected signs, where income is positive indicating higher value in high income countries, population density decreases the value of ecosystem services in this case, presumably due to congestion effects and impact on forests, and recreation and water services are both positive compared to the omitted variable, other services (including bioprospecting and conservation values). We obtain that benefits per hectare decrease with larger Holdridge zone and that wet forests appear to have greater values than the rest of Holdridge zone.

Table 5.7 Results from the GLS model

Lnvhasite	*Coefficients*	*St. error*	*z*	*Prob. > z*
water	1.8830***	0.5035	3.74	0.000
recreation	1.9901**	0.6346	3.14	0.002
lnareasite	–0.6194*	0.3003	–2.06	0.039
lnareahz	–0.8678*	0.3435	–2.53	0.012
maindry	–1.9496*	0.8259	–2.36	0.018
mainmoist	–1.3925**	0.4468	–3.12	0.002
mainrain	–2.0996***	0.4727	–4.44	0.000
lnincome	0.3379**	0.1310	2.58	0.010
lnpop	–3.4092**	1.2533	–2.72	0.007
constant	12.8847***	2.2414	5.75	0.000

Notes: n = 84; Wald ^{2} = 409.54; Prob ^{2} = 0.000

Scaling up in the baseline scenario

Results from meta-analyses can be used to predict the behaviour of the dependent variable under different circumstances described by the explanatory variables. Studies in environmental economics have used this tool for policy analysis, as it is known for being the most accurate methodology for benefit transfer estimates (Rosenberger and Loomis 2000). The benefit transfer method consists on the estimation of the value of an environmental good at a target "policy" site using an original study undertaken for another "study site" (Navrud and Brouwer 2007; Navrud and Ready 2007). Monetary estimates can be transferred as univariate transfers, using mean or median willingness to pay values, or as value functions conditioned on explanatory variables that define the attributes of the ecological and economic setting of the study site (Wilson and Hoehn 2006; Navrud and Ready 2007). Value functions can be estimated using data from one original study (Loomis 1992), or by using a meta-analysis from several case studies on similar sites (Woodward and Wui 2001; Bateman and Jones 2003; Rosenberger and Loomis 2000). This method is gaining interest in policy implementations, since it is less costly than primary valuation studies. However, the reliability of the method and transferability procedures are still under discussion in the literature (Wilson and Hoehn 2006; Moeltner *et al.* 2007; Johnston and Duke 2010; Johnston and Thomassin 2010), and further analysis are needed to explore the accuracy of the benefit transfer estimates, where this chapter provides such an example.

Scaling up is a form of benefit transfer that is commonly applied in meta-analyses (de Groot *et al.* 2012; Ghermandi and Nunes 2013). In this case, we will scale up our value estimates to obtain the aggregated ecosystem services values for the different countries we are analyzing. Box 5.5 provides a review of the advantages and challenges of benefit transfer for policy purposes.

Box 5.5 Policy advantages and challenges of using benefit transfer techniques from meta-analysis

Policy advantages:

- All methods of benefit transfer represent cheaper and quicker alternatives to primary valuation.
- Some forms of benefit transfer require less sophisticated econometric skills, which facilitates the use by non-experts. This is possible when given a benefit transfer function.
- Benefit transfer from meta-analyses (in contrast to other benefit transfer techniques) compile extensive evidence on the value of the ecosystem services under valuation.

Policy challenges:

- Estimations based on the benefit transfer method can suffer from significant "transfer errors." One way of reducing them is controlling for the highest number of variables.
- Most of the published literature on monetary values of ecosystem services is based on best-fit-principles which often include very site-specific explanatory variables, rather than core variables in line with economic theory, making the transfer across sites more difficult and, sometimes, less reliable.

Scaling-up function and Holdridge zones areas

From the results of the meta-regressions we construct a meta-function for the benefit transfer and scaling up, following previous work on the field (e.g. Richardson and Loomis 2009; Martin-Ortega *et al.* 2012):

$$\begin{aligned} \ln \textit{value} &= \textit{const} + (1.88 \times \textit{water}) + (1.99 \times \textit{recr}) - (0.62 \times \ln \textit{area}) \\ &- (0.86 \times \ln \textit{hz}) - (1.94 \times \textit{dry}) - (1.39 \times \textit{moist}) - (2.09 \times \textit{rain}) \\ &+ (0.33 \times \ln \textit{income}) - \ln \textit{pop} \end{aligned} \tag{5.2}$$

with the benefit transfer function in equation 5.2, we are able to substitute the areas and Holdridge zones types for the different countries, and estimate the range of benefits from water and recreation services at present. Values decrease marginally with the size of forest indicating a decreasing value per hectare to additional hectares, as previous work has found (Barrio and Loureiro 2010; Ojea *et al.* 2010, among others).

Using STATA command *predict* and the hectares of Holdridge zones for the baseline scenario we are able to estimate the values per hectare and Holdridge zones using our function (equation 5.2). Note that our dependent variable is in its

logarithmic form and we need to conduct a transformation to convert it to $/ha. Following Cameron and Trivedi (2009), we apply the conversion using the following formula in equation 5.3 in order to avoid errors of directly taking the exponential:

$$yhatnormal = \exp(lyhat) \times \exp(0.5 \times (yhat\_se^2)) \qquad (5.3)$$

where *yhatnormal* is the correct predicted value in $/ha.year, *lyhat* is the logarithm of the predicted value per hectare per year (*yhat*), and *yhat_se* are the standard errors of the model.

As an input for the scaling up we are using the Holdridge zones areas and types for the countries where we want to estimate benefits for. These refer to year 2000 as our baseline scenario and are summarized in Figure 5.2.

Note that in most countries moist forests are the most abundant, followed by dry and wet forests. Rain forests have a very small area, and only appear in several countries (Nicaragua, Honduras, Guatemala, Costa Rica and El Salvador). This distribution together with the explanatory variables from the meta-analysis are determining ecosystem services values under the current baseline year (2000).

Scaling up results

Marginal values for water and recreation ecosystem services estimated for given Holdridge zones areas and scaled up by Holdridge zones types in the baseline scenario are shown in Tables 5.8 and 5.9 respectively. These are values per Holdridge zones type and the range corresponds to the variability of the estimates among countries. We have included in another column estimates from previous

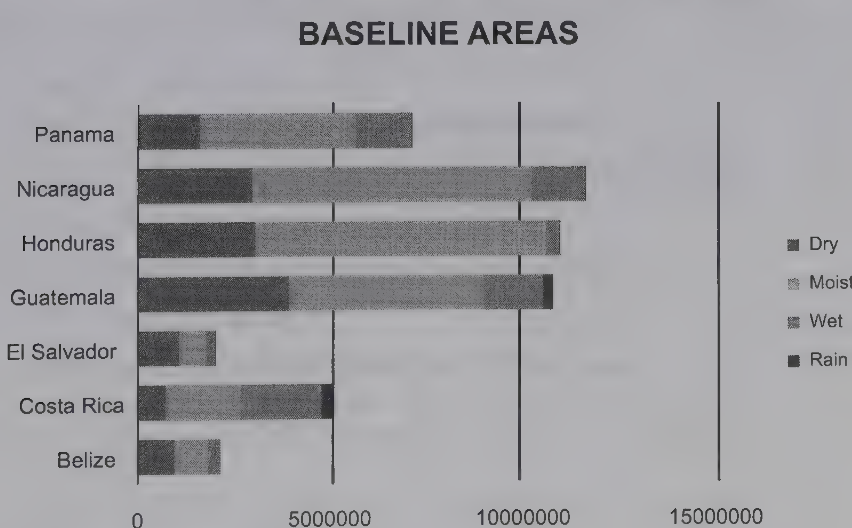

Figure 5.2 Holdridge zone areas for Central American countries in the baseline scenario

studies on the same region in order to compare our results, observing that these are fitting low bound ranges for dry moist and wet forests, while for rain forests values are on the upper bound. The baseline values are also presented in maps in the following pages, showing the spatial distribution of these values in Central America. Note the darker the color, the higher the benefits per hectare.

Table 5.8 Average values per hectare and year for water services in the baseline scenario

Water	*Average*	*Range*	*Literature*
Dry	0.68 $/ha.yr	0.50–0.92	4–238 $/ha.yr Torras (2000)
Moist	1.58 $/ha.yr	0.27–2.20	6–245 $/ha.yr Costanza *et al.* (1997)
Wet	9.78 $/ha.yr	6.54–15.08	15–850 $/ha.yr Thompson *et al.* (2009)
Rain	877.44 $/ha.yr	1.93–5060.69	

For water services we obtain that rain forests are associated with higher per hectare values. The large range is due to the scarcity of rainforests in some of the countries, where a small number of hectares is found to have very large value. Wet forests are also valued higher than moist and dry forests. In the case of recreational services, we see that values per hectare are greater than those for waters services. Also correspondingly, rain and wet forests have the highest benefits per hectare associated.

Table 5.9 Average values per hectare and year for recreation services in the baseline scenario

Recreation	*Average*	*Range*	*Literature*
Dry	1.15 $/ha.yr	0.89–1.50	2–470 $/ha.yr Thompson *et al.* (2009)
Wet	2.70 $/ha.yr	2.33–3.49	37 $/ha.yr Torras (2000)
Moist	14.61 $/ha.yr	10.35–21.35	5–10 $/ha.yr Pearce (1998)
Rain	781.23 $/ha.yr	2.72–4526.86	112–36 $/ha.yr Costanza *et al.* (1997)

Aggregated values for the baseline scenario

The aggregated benefits per country per year can be calculated from the values per country and per Holdridge zone, and the total area of the Holdridge zones at the baseline scenario. Table 5.10 shows the aggregated values per hectare per year for the services studied.

If we look at the aggregated values from water and recreation per country, we can observe how the greatest values for rain forests appear in countries with the smaller rainforest areas (El Salvador). We also observe that, for some countries, values for wet forests are higher than values for other Holdridge zones The scaling-

Table 5.10 Results for the annual aggregated benefits per country and Holdridge Zone in the baseline (2000)

M$/yr	*Belize*	*Costa Rica*	*El Salvador*	*Guatemala*	*Honduras*	*Nicaragua*	*Panama*
Water aggregated (M$)							
Dry	0.76	0.65	0.82	1.92	1.61	1.57	1.04
Moist	1.75	3.09	1.50	6.70	9.57	9.23	5.53
Wet	4.10	13.38	3.57	10.88	4.35	10.05	10.32
Rain	–	0.51	0.43	0.49	0.19	0.21	0.21
Total	6.61	17.63	6.32	19.99	15.71	21.06	17.10
Recreation aggregated (M$)							
Dry	0.26	0.19	0.02	1.02	0.39	1.02	0.20
Moist	0.58	0.92	0.03	3.52	2.42	6.19	1.12
Wet	1.22	3.79	0.07	5.07	0.87	5.81	1.86
Rain	–	0.13	0.01	0.20	0.03	0.09	0.03
Total	2.05	5.02	0.13	9.82	3.71	13.11	3.22

up exercise allows to estimate more disaggregated results, with values per country, service and Holdridge zone. In Figures 5.3 and 5.4 we can see how the aggregated benefits are distributed in countries and Holdridge zones, obtaining that wet forests are the ones providing the greater share of economic benefits.

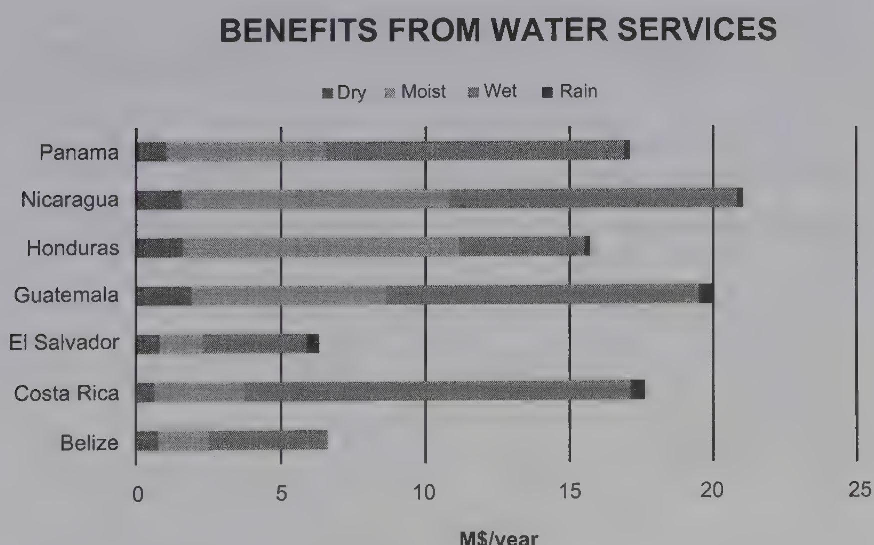

Figure 5.3 Aggregated benefits for water services in Central America in the baseline scenario

In the case of recreation, the aggregation has been done taking into account the per centage of protected areas on each country, as obtained from United Nations Development Programme statistics (Table 5.11). This is necessary since recreation is a service that is enjoyed on specific sites, and recreational forests are suitable for it. Not all forests are accessible and suitable for recreation, so our analysis has focused on protected areas. This is the main reason why the aggregated benefits from recreation are slightly smaller than for water services, which depend on all the watershed area (knowing that benefits per hectare were greater).

Table 5.11 Protected areas per country (2010)

Country	*Percentage protected areas*
Belize	20.59
Costa Rica	17.64
El Salvador	1.38
Guatemala	29.51
Honduras	13.86
Nicaragua	36.84
Panama	11.49

Source: United Nations Development Programme.

Aggregated annual benefits for recreation are shown in Figure 5.4. Nicaragua and Guatemala are the two countries with higher aggregated annual benefits, derived from both recreational and water services. However, it is Panama and Costa Rica

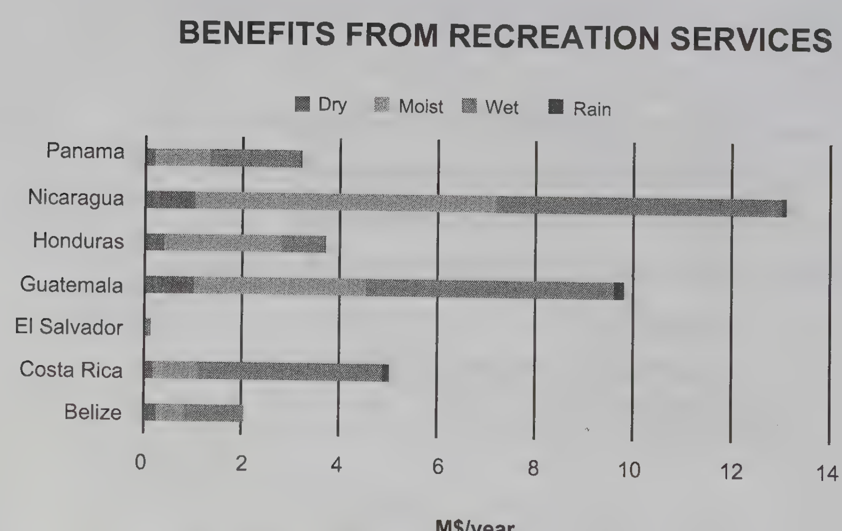

Figure 5.4 Aggregated benefits for recreation services in Central America in the baseline scenario

having greater water benefits from tropical forests, and Costa Rica and Honduras for recreation services.

Water and recreation values under climate change

We have so far developed an estimation of the benefits obtained from water and recreation ecosystem services in a baseline scenario, where forest types and areas correspond to the current environmental conditions. In this section, the analysis focuses on different scenarios obtained from the projected impacts of climate change on the vegetation model. We first present the climate change scenarios on which we based our results, and second we estimate the economic values for these scenarios.

Climate change scenarios

In this section we replicate the methodology used in the scaling up process but with inputs from the expected Holdridge zone distributions under future climate change. We projected forest types under 2100 climate modelled by ECHAM5/MPI-OM general circulation model under high (A2, three simulations), intermediate (A1B, four simulations) and low (B1, three simulations) emission scenarios. ECHAM5 has also been selected by ECLAC for a report on the economic impacts of climate change in Central America (Lennox 2010) and provides for mean climate anomalies relative to other global climate models (GCMs) available. Since each emission scenario has several GCM simulations for which Holdridge zones were projected, we averaged the Holdridge zone areas for each emission scenario.

Future benefits from forest ecosystem services in our model therefore are only dependent on the potential Holdridge zones areas and types and how values depend on them according to the meta-analysis. We do not include any other projected variable, such as income or population since these are already part of the climate model that we have used for predicting vegetation changes. Further analysis will be needed to provide evidence on how economic projections can play an important role in future ecosystem services values.

A2, A1B and B1 scenarios from the IPCC were chosen as they represent a range of potential pathways for economic and environment development and their associated greenhouse emissions. The transference of benefits using the meta-function obtained in equation 5.2 is based on the expected shifts in Holdridge zones under climate change. Holdridge zones are expected to change in distribution considerably, although no significant loss of forests to other Holdridge zones is expected. Note that we are considering potential Holdridge zones, both in the baseline and in the future scenarios for reasons of comparability. We are interested in the direction and relative change in benefits, as we have no accurate information on the uses of land in such a far prospect of 2100.

Figure 5.5, shows how Holdridge zones are projected to change under climate change, from baseline year 2000 to future scenarios A2, A1B and B1 in year 2100.

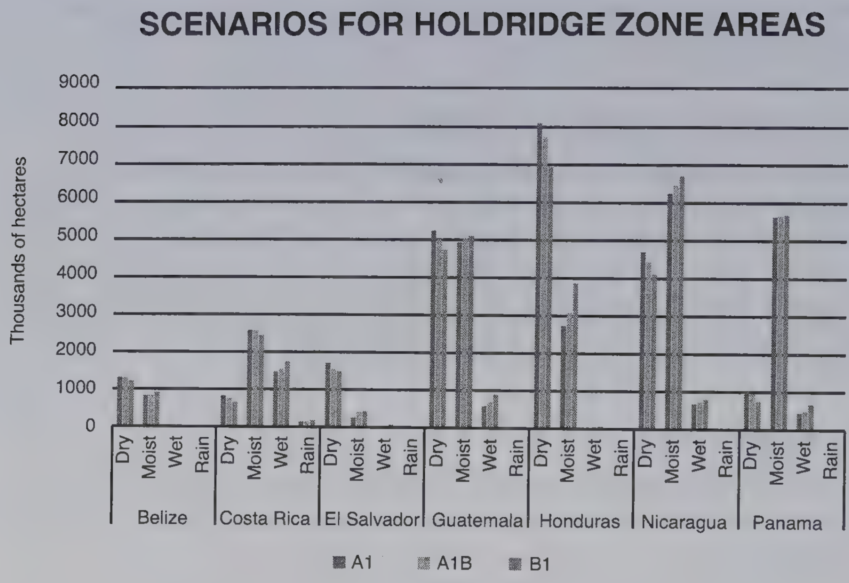

Figure 5.5 Holdridge zone areas in baseline (2000) and future climate scenarios (2100)

In most cases we observe an increase in dry forests respect to the baseline, and a dramatic decrease in rain forests (where these are expected to disappear in several countries) as well as on wet forests. For wet forests results show different evolutions for southern and northern countries, where moist forests increase in Costa Rica and Panama and decrease in the rest of countries. There are also some differences between scenarios of climate change, where scenario B1 shows the less dramatic changes in Holdridge zones.

Results for future climate change scenarios

We run estimations of ecosystem services values using our economic model (equation 5.2) and the potential Holdridge zones area for future climate scenarios (Figure 5.2). As a result we are able to obtain the values per hectare per year for

Table 5.12 Values per hectare per year for water services under future Holdridge Zones

Water	*Scenario A2*		*Scenario A1B*		*Scenario B1*	
$/ha.yr	*Average*	*Range*	*Average*	*Range*	*Average*	*Range*
Dry	0.62	0.42–0.86	0.64	0.43–0.90	0.66	0.43–0.95
Moist	1.76	1.30–3.35	1.66	1.29–2.70	1.64	1.29–2.64
Wet	146.71	7.11–856.32	78.97	7.00–421.05	37.84	6.77–166.33
Rain	96.59	2.75–163.21	46.42	2.64–105.27	14.50	2.45–34.11

both services, which are expected under future conditions for forest distribution. Water values under climate change scenarios are included in Table 5.12, while recreation value estimates are included in Table 5.13.

Economic values for water services are greater for wet forests, followed by rain, moist and dry forests. The decrease on the value per hectare of rain forests can be explained by the disappearance of rain forests in those countries where it was very scarce in the baseline, and therefore raising its value (this is the case of El Salvador). The more sustainable scenarios are related to lower values per hectare, and this may be due to the same reason of less scarce Holdridge zones. For recreation, the results show the same patterns as for water, and values per hectare are slightly higher in wet and rain forests but smaller in moist forests, as compared with water services.

Table 5.13 Values per hectare per year for recreation services under future Holdridge Zones

Recreation	*Scenario A2*		*Scenario A1B*		*Scenario B1*	
\$/ha.yr	*Average*	*Range*	*Average*	*Range*	*Average*	*Range*
Dry	0.49	0.20 –0.86	0.51	0.20–0.93	0.53	0.20–1.04
Moist	0.91	0.14–2.43	0.78	0.14–1.72	0.77	0.14–1.73
Wet	238.66	1.85–1380.41	139.60	1.60–771.42	63.49	1.28–334.32
Rain	77.94	0.00–212.67	47.17	4.66–82.81	15.92	4.16–31.96

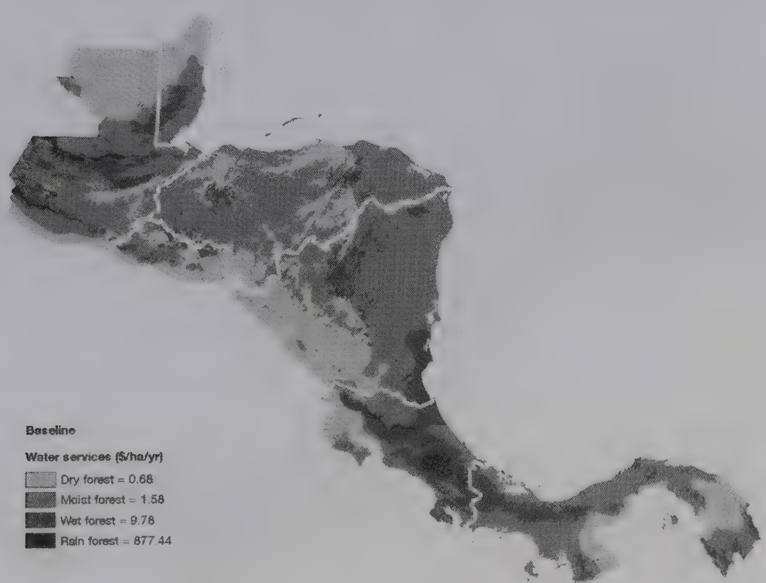

Figure 5.6 Distribution of water estimated benefits in the baseline scenario

Figures 5.6, 5.7, 5.8 and 5.9 show the distribution of values for water services under consideration of each of the scenarios (baseline, A2, A1B and B1). Figures 5.10, 5.11, 5.12 and 5.13 show the distribution of values for recreation services in

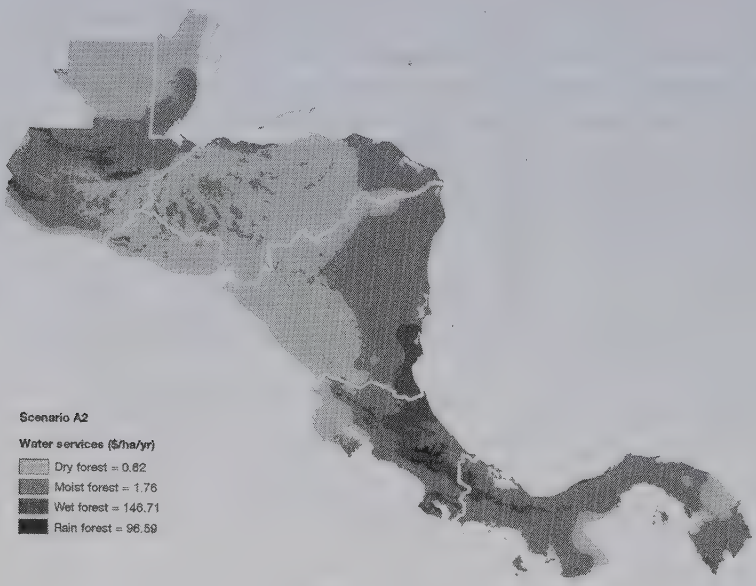

Figure 5.7 Distribution of water estimated benefits in scenario A2

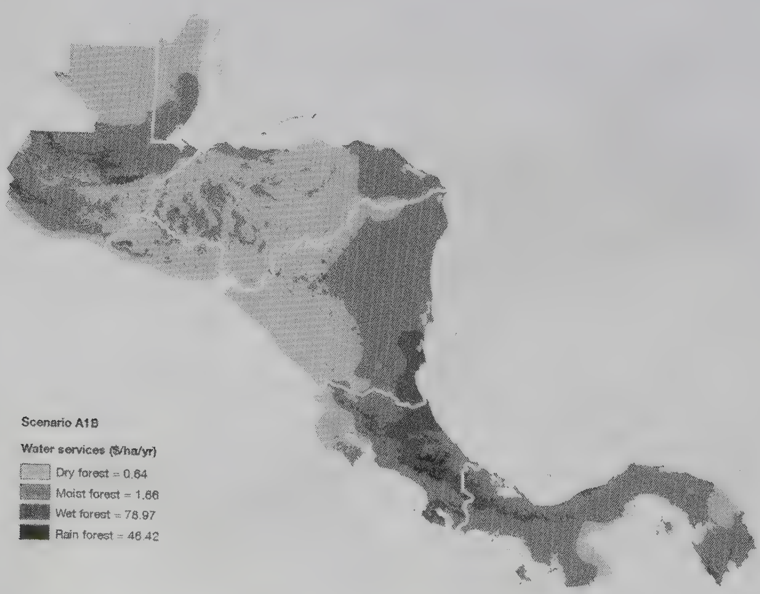

Figure 5.8 Distribution of water estimated benefits in scenario A1B

the same climate scenarios (baseline, A2, A1B and B1). The ascending tonal gradient corresponds to increasing values (i.e. darker zones correspond to higher monetary values in $ per hectare per year).

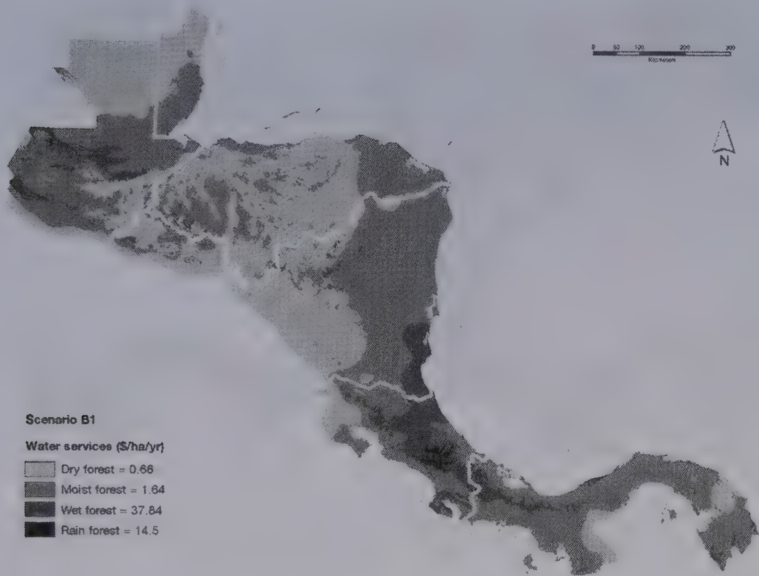

Figure 5.9 Distribution of water estimated benefits in scenario B1

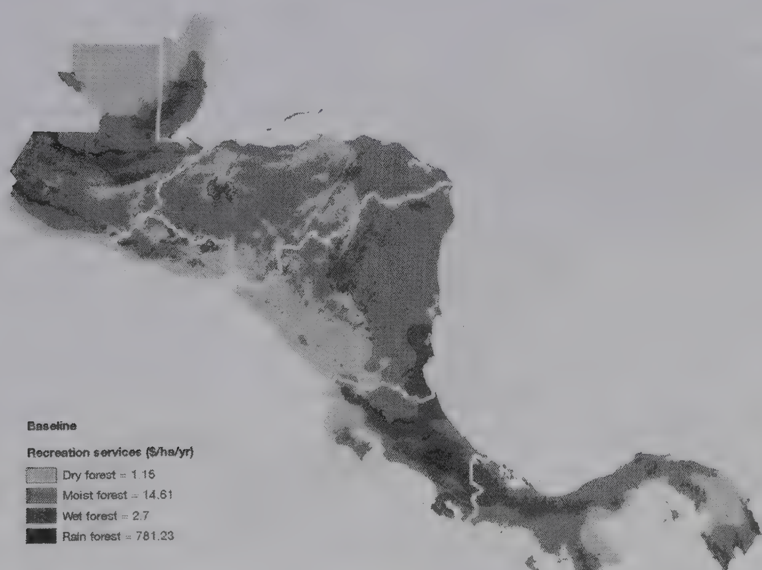

Figure 5.10 Distribution of recreation estimated benefits in the baseline scenario

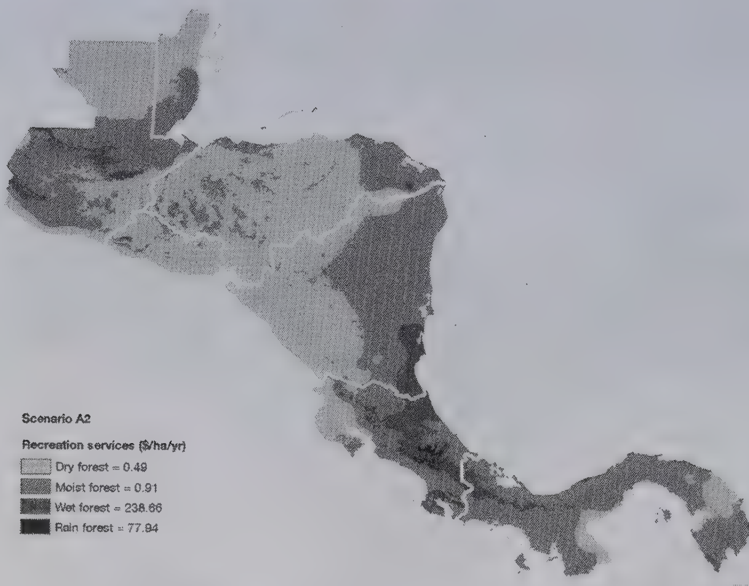

Figure 5.11 Distribution of recreation estimated benefits in scenario A2

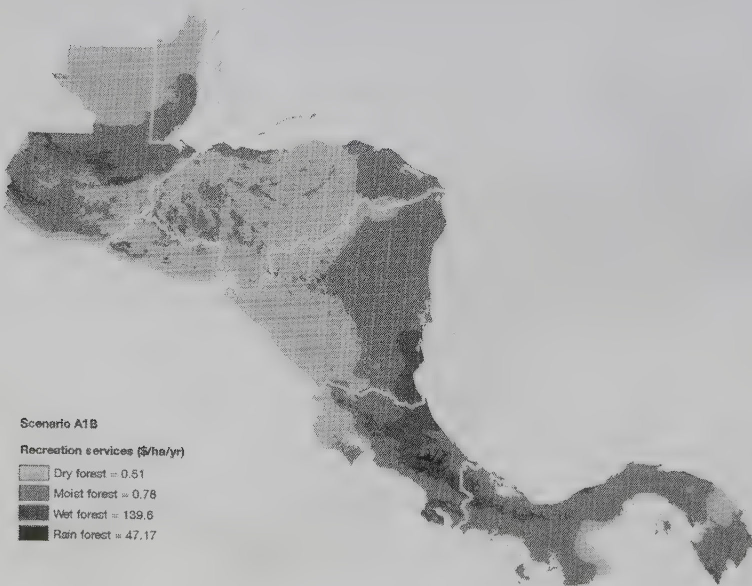

Figure 5.12 Distribution of recreation estimated benefits in scenario A1B

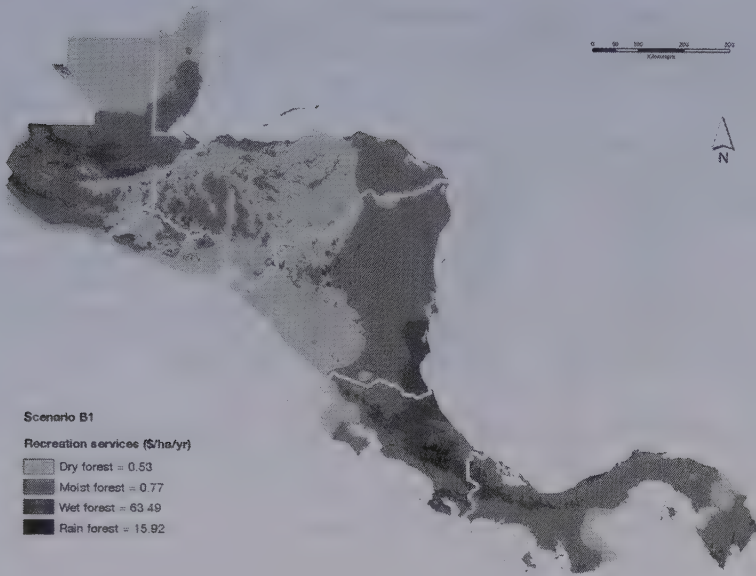

Figure 5.13 Distribution of recreation estimated benefits in scenario B1

Results

Future values of ecosystem service

First we calculate the aggregated flows of ecosystem services benefits for each future scenario by substituting Holdridge zones areas and types into our economic model. Figure 5.14 shows the results for the aggregated estimates, with the expected benefits per country for each Holdridge zone under the three climate scenarios, for water and recreation services.

From the graph we can easily observe how the aggregated benefits for water services are greater than for recreation, the main reason is that recreation is not provided by all forest area, as opposed to water services. For all the cases and future scenarios, we observe a clear net loss in annual benefits from ecosystem services as compared to the current flows shown in the baseline. Honduras and Guatemala appear to have major losses in water services, while Guatemala and Nicaragua are expected to lose a lot in recreation service flows. We remind that these benefit flows are the result of considering the potential Holdridge areas and that, in a real situation, we would had a portion of these potential forests converted to other non-productive or less productive uses.

Economic impacts of climate change on water and recreation

Last step of the research is to estimate the expected impacts of climate change on the benefits obtained from water and recreation services in Central American

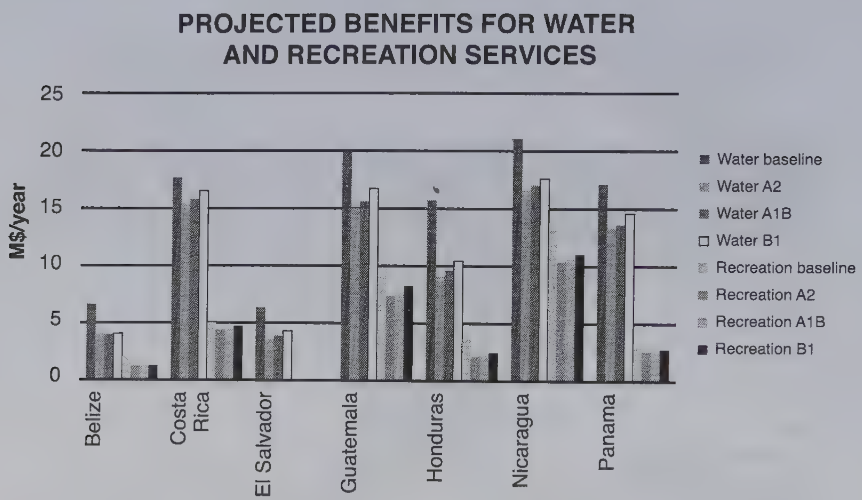

Figure 5.14 Comparison of the aggregated benefits across services and scenarios (in M$/year)

forests. We have already calculated the benefits today and under different future scenarios for Holdridge zones. In this last step we proceed with the comparison to obtain the different economic impacts. Results are provided and discussed in relative terms, as the economic impacts are better understood in this way, given that we are comparing values from potential vegetation types, which are overestimating the benefits that are obtained in a real scenario.

We can see graphically how these changes in benefits take place. Figures 5.15–17 show the impacts on water and recreation services under scenarios A2, B1 and A1B respectively. The relative change in benefits gives us an idea of the proportion and magnitude of change, without having to rely on absolute quantities that are based on potential forest areas. The bars represent the estimated change in annual benefits obtained from water and recreational services in the different countries and for all types of forests. We observe that relative losses are very high, with as much as 100 per cent rain forests disappearing in countries such as El Salvador or Honduras. Honduras has a very big increase in dry forests, above 88 per cent in the best case scenario. In general terms, recreation suffers bigger relative losses than water services, probably due to the sensitivity of visitors to the wet and rain forests, although the relative results are very similar due to their dependence on forest types and areas. The main differences between recreation and water services appear for the moist forests, where recreation values may be important compared to water service values. In absolute terms however water services are expected to cause the bigger losses. We can observe how dry forests are expected to increase annual benefit flows but wetter Holdridge zones are expected to decrease. The same result stands for recreation. Note also that the expected changes in the flow of economic benefits is not always a negative one, and that positive impacts are expected in dry forests for many of the countries, both for water and recreation services. However, these gains in dry and in some cases moist

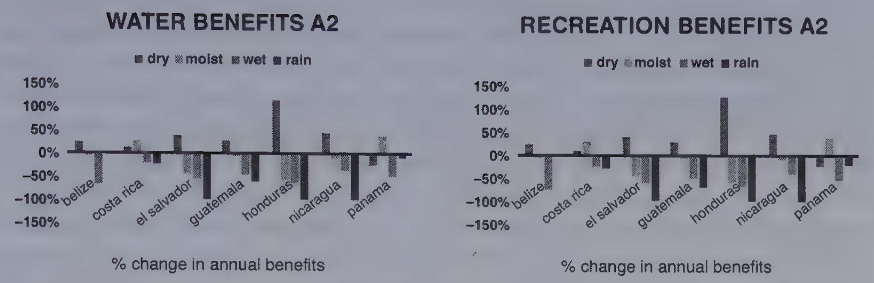

Figure 5.15 Climate Economic impacts in water and recreation services in scenario A2 (%)

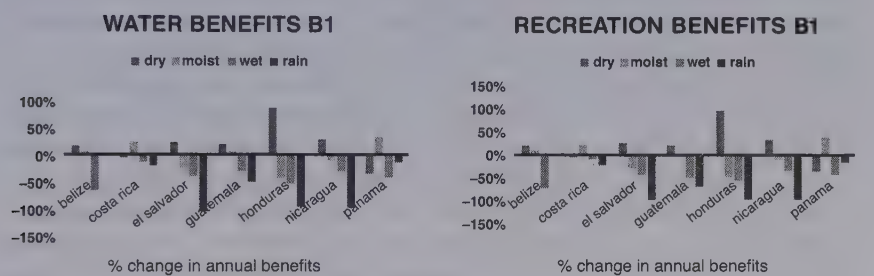

Figure 5.16 Climate Economic impacts in water and recreation services in scenario B1 (%)

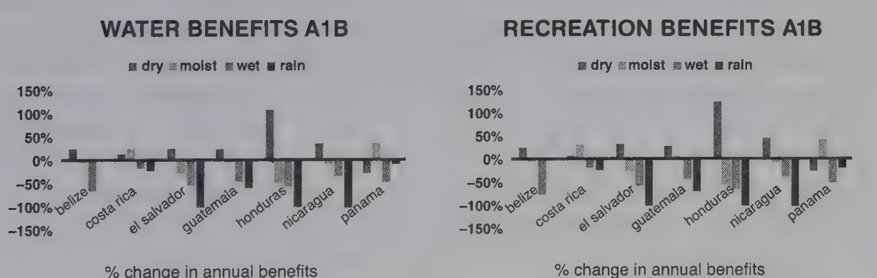

Figure 5.17 Climate Economic impacts in water and recreation services in scenario A1B (%)

forests clearly do not outperform the losses and this exercise evidences the magnitude of the losses of forest service's values due to climate change.

Conclusions

The present study provides a new contribution to the environmental economics literature, specifically to the estimation of the economic impacts of climate change on ecosystem services. The methodology developed here has some novel aspects.

First is that we have conducted a regional scale meta-analysis based on case studies focused solely on tropical forests of Central and South America. Our objective has been to have a collection of studies where the environmental good, in this case forests, is consistent among sites, by sharing common climatic and biological features. With a few exceptions, meta-analyses in forest services have focused on large scales where different habitats and bio-geographical regions are included. In this exercise we focus on tropical forests in a group of countries with similar bio-geographical features. This way, we are able to conduct benefit transfer among similar habitats and countries in order to avoid the so called generalization problem (Rosenberger and Stanley 2006). Second contribution is that we specifically obtain biophysical information at the study site level, both the area and the forest type, while previous meta-analysis have used ecological variables at the country level or without differentiating vegetation types. Finally, we have estimated flows of benefits instead of stocks, meaning that the flow of benefits from ecosystem services annually obtained is the focus of the analysis. The ecosystem service approach favors the maintenance of the flows of services and we have taken such an approach. These flow values are dependent on the forest area and type, represented by the vegetation or Holdridge zones. Additionally, a relevant contribution of this exercise is that we employ benefit transfer in the context of climate change, which supposes an example of an approach that, despite its limitations, can have important policy uses and applications. To our knowledge, only a few studies have made attempts in this direction (e.g. Ding *et al.* 2010 for European forests and Chiabai *et al.* 2011 for global forests). These studies are based in forest biomes and the impacts expected from climate change, using climate and vegetation models. The inclusion of Holdridge zones supposes a downscaling in the vegetation models where we are able to link specific sites for which we have economic value estimates with the existing and projected potential vegetation types. By increasing the accuracy of the biophysical variables, the present work contributes to the analysis of climate change economic impacts on ecosystem services at a finer scale. Although future research will have to keep working along these lines in order to improve the accuracy and reduce uncertainty in the estimation of economic impacts from climate change. Finally, our exercise includes non-market values for water and recreation ecosystem services for which we provide an estimation of the impacts in services that are not part of the traditional or accountable economy (for example, they are not collected in the GDP). Addressing non-market ecosystem services is challenging since there are not references in existing markets and the literature on valuation remains the only source for this information.

From the consecution of this exercise we have obtained an estimation of climate change economic impacts based on bio-physical data, through the interactions between ecosystem characteristics, ecosystem services and benefits. The results can be interpreted at four levels:

- the forest type;
- the environmental service;
- the countries; and
- the climate scenarios.

If we look at the *forest type* or *Holdridge zones*, we find that rain and wet forests have the higher per hectare values in the present baseline scenario. Rain forests are providing the greater per hectare benefits due to its current scarcity, but its value decreases in the future due to its disappearance in some countries. Wet and moist forests are expected to have the greater values in the future as they are being heavily impacted by climate change. Dry forests increase in area under climate change with low values associated.

If we look at the *environmental services* at stake, we observe that per hectare values of recreation are higher than for water in the present, however, aggregated benefits at the country are greater for water services. Important economic losses are projected for both services with similar magnitude and direction (~50 per cent losses in wet forests), but impacts depend on forest types where losses take place in dry, wet and the majority of moist forests and dry forests present gains in benefits.

Looking at the *countries* under consideration, Guatemala and Nicaragua are generating higher benefits in the baseline, due to their large area and forest types (wet and rain forest). Costa Rica generates the greatest benefits with its wet forests. With climate change, bigger losses happen in Honduras (water) and Nicaragua (recreation).

For the *climatic scenarios*, it is clear how the more sustainable oriented scenario gives the smaller economic impacts. However, the economic impact is different for different scenarios but does not have abrupt variations, as the difference between scenarios is not large. Box 5.6 summarizes the main messages for policy applications of the present methodology based on the results we have obtained and the experience of this exercise.

Box 5.6 Challenges for policy applications of the current analysis

Future applications of this methodology should consider the following issues:

- Regional consistence: the area of study should have similar habitats and ecosystems providing ecosystem services through the same mechanisms in order to conduct a study like this one.
- Transferability: countries should also have similar economic levels to allow for benefit transfer and scaling up values.
- Biophysical units: the flow of ecosystem services should be understood and characterized by a mechanism related to known biophysical units, such as the forest type and size, water runoff, number of visits etc. Important attention should be made to non linarites in the provision of ecosystem services.
- Climate change scenarios: climate scenarios are translated into biophysical impact scenarios in this case through the potential vegetation in Holdridge zones. Additional possibilities including land use change would benefit future analyses.

This study has also some limitations that we want to acknowledge here as well. We use potential areas of forests and thus our analysis cannot be taken to understand the magnitude of the flows of benefits, neither at present or in the future, as land uses are not included in our estimates. Since we are interested in the change in ecosystem services due to climate change, the comparison of present and future potential vegetation types allows us to respond to that question. Land use has not being included mainly due to the fact that we lack information about future land uses and for year 2100 any prospect would be under a very high level of uncertainty. But this also means that we cannot see from our analysis if we should expect the area of forests disappearing in the future, for example. The second main limitation is that we constrain our economic impacts to the effect of climate change through Holdridge zones, and we do not take into consideration other economic variables such as income or population changes. Economic storylines for the next hundred years are however included in the climate projection scenarios from the IPCC, that we are employing here for the vegetation types, but a question remains open in whether we could have included population and income projections into our benefit transfer function. Third and last limitation is that we focus on one regression model for the two services, water and recreation, where a worth exploring option for future analyses would be constructing a different model per service type. This way, we would be able to track the interactions of forest types and services that are not decoupled in the present analysis. Being able to do this, however, requires a larger sample of case studies that was not possible for the present area of study.

Tropical forests are one of the most important worldwide ecosystems, both for the conservation of biodiversity and for fighting climate change. The study conducted here evidences its critical situation under current climate change projections and claims for the urgent need for action. We have shown how projected changes in climate can derive into important changes and impacts on the economic benefits obtained from ecosystem services. These impacts vary among countries and forest types, but in general terms we expect a net loss in water and recreation services in all countries due to climate change. Wet and rain forests are expected to suffer the bigger losses in area and in economic revenues, with important regional exceptions. Environmentally oriented climate scenario B1 produces the lower impacts, although their magnitude is considerably large. This means that even if societies and economies follow the most sustainable path in development, impacts on ecosystem services are severe. Therefore, more mitigation action to stop deforestation and more adaptation efforts to maintain the flows of ecosystem services are needed in the context of Central American forests. We demonstrate the magnitude of the economic impacts of impacted ecosystem services provided by tropical forests. Specifically we can see the economic importance of water and recreation services in forest that are internationally recognized mainly by carbon and biodiversity values. The value found for water and recreation services at present highlights the need to develop strategies to maintain their conservation in the face of a changing climate, and incorporate their sustainability in adaptation and mitigation initiatives, such as REDD+

(Reducing Emissions from Deforestation and Forest Degradation). Adaptation to climate change by maintaining the flows of ecosystem services is not only important for sustaining local livelihoods, but for the entire country as we observe national scale impacts to be significant. The results obtained here point to the need to target wet and rain forests, in general terms, for conservation and sustainable use, as they face heavier threats and carry the highest values in terms of ecosystem services and climate change.

The methodology presented here has proven useful for the estimation of climate economic impacts on ecosystem services and there is room for improvements. We present this exercise as a first step from which future applications should develop further refinements. The study also evidences the importance of a detailed analysis of economic impacts of climate change, which has to be consistent with the work of natural and economic scientists and bridge the gap between disciplines to answer big questions such as how climate change will affect benefits from ecosystem services. We have seen how Holdridge zones serve as a starting point to link natural and economic impacts and see the detail of ecological and economic variables. Further research should explore other alternatives for the biophysical variables of change and maybe incorporate population, income and land use scenarios that will help to understand better the economic implications of climate change.

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Notes

1 Holdridge zones are potential vegetation types determined with climatic variables (Holdridge 1947, 1967), See the "Economic impacts of climate change on water and recreation" section below for a more detailed analysis.

2 It should be mentioned that evidence suggests that not all payments for environmental services are the outcomes of a market dynamic (Martin-Ortega *et al*. 2013).

6 Cost–benefit analysis of alternative land-use scenarios

A sustainability study for the Volcanic Central Talamanca Biological Corridor

Helen Ding, Aline Chiabai and Diego Tobar

Introduction

Agricultural production is among the main drivers that are responsible for biodiversity losses, deforestation and soil degradation in Central America. Until recently, however, conservation of biodiversity and agricultural production have been considered as incompatible goals for the regional sustainable development. According to Macfadyen *et al.* (2012), measures aiming at protecting biodiversity can facilitate the preservation of ecosystem services, while the opposite is not necessarily true. Indeed, biodiversity supports ecosystem functioning which in turn sustains the provision of ecosystem services. In line with this concept, today and more than ever, the conservation of biodiversity in protected areas is becoming integrated in the agricultural matrix to provide additional services.

Current research has shown that good land-use management practices in the tropical forests and agricultural land, such as by establishing silvopastoral systems (SPS) and agroforestry systems (AGS) instead of treeless agricultural land, can provide an important solution to mitigate climate change through the increased carbon stocks in living biomass and soils (Daily 1997; Polasky *et al.* 2011; Pagiola *et al.* 2004). In addition, SPS involve the integration of woody perennials (shrubs and trees) with animal and pasture components, and have the potential to reduce the impact of livestock systems on the environment in the long term and enhance both livestock productivity and rural livelihoods (Pezo and Ibrahim 1997). A study by Harvey and Haber (1999) also shows that within fragmented forest landscapes, farm trees may represent critical habitats and corridors for plant and animal species, help maintain local and regional biodiversity and improve the ecosystem services. Given these multiple co-benefits, promoting SPS has become a very popular land-use management alternative in Central America. Moreover, it is obvious that forest conservation, SPS and other land-use practices and management may have significant impacts on the future economic returns to people whose livelihoods rely on the utilization or commercialization of natural resources. Therefore, a good understanding of the economic

and ecological impacts of land-use changes and management is essential for designing cost-effective management strategies to exploit sustainable agricultural practices.

The current chapter proposes a methodological framework for a cost–benefit analysis of alternative land-use options, and illustrates its application in a specific site in Costa Rica, known as an important biodiversity hotspot, the Central Talamanca Volcanic Biological Corridor (VCTBC), located in Cartago province. This site has been chosen for its high ecological and cultural value, due to its proximity to the Caribbean region, and secondly because of the existence of conflicting interests and trade-offs between the protection of the environment and current economic activities. Due to these reasons, it represents a very interesting location for analyzing the competing uses of land.

The area extends over 72,000 hectares, spanning 18 districts of Cartago province. Forty per cent of this area is covered by forests that protect significant resources, among them local biodiversity and water. It is home to 889 species of vertebrates (59 per cent of the national total), 601 species of birds (70 per cent), 169 species of mammals (70 per cent), 73 species of reptiles (31 per cent), and 46 species of amphibians (25 per cent). The VCTBC is a part of the Central American Corridor implemented in Costa Rica to allow wildlife to move from one protected area to another thanks to the connectivity provided by remaining forests and afforestation efforts in some farming areas.

In addition to forest conservation, dominant economic activities of the region are agricultural production, including mostly coffee, sugarcane and livestock. By 2010, the land uses within the corridor embraced 51 per cent of forest cover, 31 per cent of agricultural fields (pastures, coffee, sugarcane and forest plantations), 1 per cent of urban settlements, and 6 per cent of secondary semi-natural shrubby vegetation. Other important activities that may affect the regional sustainable economic development include trade and growing tourism (mainly ecological and adventure tourism). Collectively, these economic activities have resulted in the reduction and fragmentation of forest cover and the creation of complex mosaics of small agricultural plots, pastures, fallows, secondary growth and forests, in which the diversity and composition of plant and animal communities is often degraded (Harvey *et al.* 2005). Moreover, continual increase in global temperature will only accelerate the alterations in the functioning and performance of the tropical forest ecosystems, and ultimately affect human livelihoods through changes in the flows of ecosystem goods and services.

In order to improve the land-use management in the VCTBC and increase the regional resilience to climate change, four management principles have been established, including (i) regional institutional coordination for management, funding, systematization of experiences and data within the corridor, (ii) promotion of activities and capacity building related to environmentally friendly agricultural production, (iii) development of biodiversity conservation and research activities within the corridor, and (iv) use of education and diffusion of information as a strategic tool for fostering growth and consolidation of the corridor (Canet-Desanti *et al.* 2008).

Under such circumstances, this chapter develops a methodological framework to assess costs and benefits (including non-market values) associated with changes in the extent and composition of tropical forests, grasslands and croplands in the context of climate change, and to evaluate the trade-offs and synergies of conservation strategies for biodiversity conservation and provision of ecosystem services.

In particular, the physical indicators of a land-use change model are incorporated in a cost–benefit framework to project the policy-driven and economic land-use changes in the VCTBC from 2010 to 2030. As a consequence, changes in the flows of ecosystem services can be first evaluated in biophysical terms under different land-use change scenarios and then translated into economic magnitudes. The proposed approach is designed to assist local policy-makers to identify optimal land-use management strategy that is able to both generate the highest long-term co-benefits and incorporate the interests of different stakeholders.

This chapter is organized as follows. Below we present the theoretical background of the social cost–benefit analysis (SCBA), the formulated net present value (NPV) calculation for the case study area and the land-use change model. The following section integrates the results of the land-use change model in the SCBA to evaluate three scenarios of future land-use changes in VCTBC. We then discuss the preliminary CBA results and perform a sensitivity analysis. Finally, conclusions are drawn and we provide some policy recommendations derived from the CBA.

The methodology: social cost–benefit analysis

A theoretical background

The cost–benefit analysis (CBA) is an economic decision-making approach, consisting of a set of procedures for defining and comparing benefits and costs (Zerbe and Dively 1994). CBA is used in the assessment of whether a proposed project, program or policy is worth doing, or to choose between several alternative ones. It involves comparing the total expected costs of each option against the total expected benefits, to see whether the benefits outweigh the costs. In CBA, benefits and costs are expressed in money terms and are adjusted for the time value of money, so that all flows of benefits and costs over time are expressed on a common basis in terms of their present value. Box 6.1 summarizes the standard steps to follow while conducting a CBA. In the present chapter, a CBA framework is combined with a land-use change model in order to identify and quantify the benefits and costs associated with various land-use management practices and to assess the trade-offs between economic-driven development and sustainable forest management. In this regard, the current framework goes beyond a conventional cost–benefit analysis of a forest conservation program and refers to a social cost–benefit analysis (SCBA) which incorporates social externalities resulting from alternative land-use scenarios and their impacts on local livelihoods.

Box 6.1 Ten standard steps for CBA of land-use changes in an ecosystem service framework

1. Specify the land-use change scenarios to be compared.
2. Specify the changes in ecosystem services resulting from different land uses and management practices, as well as identify the type of benefits and costs affected by the changes.
3. Choose the type of welfare measure that should be evaluated (e.g. WTP, WTA).
4. Decide whose benefits and costs should be counted (i.e. which stakeholders, what scale should we aggregate over, from local to global levels).
5. Predict the impacts quantitatively as a result of land-use changes over the next 20 years.
6. Monetize all impacts (benefits and costs).
7. Discount benefits and costs occurring at different points in time, in order to obtain present values.
8. Compute the net present value of each alternative land-use scenario, and the net present value of each land-use type within the scenario.
9. Perform sensitivity analysis on key variables, by applying different discount rates.
10. Make a recommendation based on the results

Source: adapted from Mullan and Kontoleon (2008).

In other words, the proposed SCBA involves a full assessment of the costs and benefits of land-use planning from a social planner's perspective, taking into account not only the costs and benefits accrued to the individual stakeholders in financial terms, but also some of the most important consequent externalities born by the society, in particular the benefits and costs for which no markets exist. Particular attention is also given to the distribution of costs and benefits among different (both local and international) stakeholders, so as to identify winners and losers under different land-use management planning. In this regard, the study considers some of the equity implications of the land-use management, along with its cost-efficiency. In other words, the results of the SCBA may provide insights in the development of innovative financing instruments that can efficiently reallocate resources from the winners to the losers, in order to compensate the costs of forest conservation and sustainable farming practices accrued to local stakeholders, and therefore improve the equity and optimize net social benefits associated with conservation planning (Hockley and Razafindralambo 2006).

The net present value

The net present value (NPV) is the difference between total discounted benefits and total discounted costs associated with a land-use scenario over a period of time. If the NPV is positive, the benefits outweigh the costs, meaning that the future land-use pattern can gain a net social benefit. If the NPV is negative, the land-use pattern will lead to a net economic loss to the society. In the present study, the NPV is likely to be positive in all future scenarios due to the consideration of non-market benefits provided by different landscapes. However, only the one with the higher NPV will generate a greater economic return to the regional communities in VCTBC. The calculation of the NPV is illustrated in equation 6.1:

$$NPV = \sum_{i=1}^{m}\sum_{t=0}^{n}\left[\frac{B_{i,t} \times \Delta ha_{i,t}}{(1+d)^t}\right] - \sum_{i=1}^{m}\sum_{t=0}^{n}\left[\frac{C_{i,t} \times \Delta ha_{i,t}}{(1+d)^t}\right] \tag{6.1}$$

where *NPV* is the net present value associated with the provision of ecosystem services (ES) by different types of landscapes in a given future scenario; $B_{i,t}$ are per hectare benefits based on land productivity (which refers to the total economic value per hectare) associated with land-use type *i* and related to the provision of ecosystem services in year *t*); $C_{i,t}$ are per hectare costs associated with land-use type *i* in year *t*; $\Delta ha_{i,t}$ are changes of land area for land-use type *i* in year *t*; and *d* is the discount rate.

The first half of the equation calculates the total discounted economic benefits of ES provided by a total of *m* types of landscape over a number of years *n*, at a discount rate of *d*; and the second half of the equation calculates the total discounted costs associated with the different types of landscape over a number of years *n*, at a discount rate of *d*.

In the present study, total benefits are evaluated on the basis of land productivity factors for all types of ES provided by different landscapes in a future scenario. These ES include (i) agricultural and timber products, (ii) biodiversity value, (iii) *in-situ* water supply for hydropower, (iv) carbon sequestration, (v) recreation and tourism, which provide either monetary or non-monetary benefits to the local livelihoods. For simplicity reasons, we assume a constant land productivity ($/ha) over time and a linear relationship between land-use changes *i* and values of ES. This means that under one land-use scenario, the value difference of the same type of ES between the two successive years is only due to the annual changes of the land area, whereas the productivity value per hectare is assumed to be constant with time.

In contrast, the estimation of total costs associated with the maintenance and/or changes of land uses is more straightforward. The costs include conservational costs of forest (including opportunity costs, i.e. the foregone benefits from economic activities on the land other than conserving the forests), maintenance costs of traditional croplands, and/or extra costs required for converting from traditional land uses to more sustainable land uses.

A cost–benefit analysis of alternative land-use change scenarios in VCTBC

Defining and modelling the land-use change scenarios

Land-use cover change (LUCC) is a complex phenomenon that involves interactions between social and natural systems (Geist and Lambin 2001; Lambin *et al.* 2003). LUCC such as deforestation and forest regrowth is the result of how people use the land as social, economic, and political drivers are changing. These drivers include large-scale, structural dynamics such as price shifts in international markets and changes in national development policy (Calvo-Alvarado *et al.* 2009). In Costa Rica, government policies in general and economic policies in particular have historically affected the use of forest resources (De Camino *et al.* 2000). For instance, rapid deforestation was observed in the country between 1950 (> 50 per cent of forest coverage) and 1986 (29 per cent of forest coverage), mainly owing to the conversion of forests to agricultural lands and cattle pastures to meet the increasing demand for beef in the 1960s and 1970s (Chomitz *et al.* 1999). In the 1980s, the rates of deforestation have been significantly reduced as a result of joint efforts of Structural Adjustment Programs (SAPs)[1] and Costa Rican policies.[2] Since the 1990s, a shift from the simply forest management and deforestation to the incentivized forest preservation and reduced environmental exploitation has been observed as a result of some important changes in the natural resource management laws, including a national ban that prohibits logging activities in protected forests and mountain monuments. All these efforts have resulted in steady recovery of forest areas, and by 2005 the total forest coverage has reached 51 per cent again.

Studies show that future land-use changes in VCTBC will most likely be influenced by changes in global markets and local environmental regulations. In particular, it is generally believed that:

- future trends of regional forest management will be focusing on improving the overall quality of the forest and its resilience to climate change (Brenes 2009);
- international market prices of beef and coffee are most likely to continue to increase over years (Calvo-Alvarado *et al.* 2009); and
- regardless of changes in market prices, strong regional sustainable development strategies will continue to support the conversion of traditional pastures to silvopasture and agroforestry lands, which provide a range of ecosystem-derived environmental, social and economic benefits.

In this context, we can develop three future land-use scenarios by making different assumptions on the above mentioned influencing factors to represent different mixes of economic and environmental development interests in the region. The three scenarios refer to:

- *Scenario 1*—a modest sustainability scenario (business-as-usual);
- *Scenario 2*—a strong sustainability future scenario; and
- *Scenario 3*—an intensive economic development scenario.

The key assumptions underlying the three future scenarios are summarized in Table 6.1 (see also Table 6.9 in the Annex for a more detailed description of the scenarios).

Table 6.1 Key assumption for alternative future scenarios description (compared with baseline year)

Assumptions on key aspects	*S1: The modest sustainable development development future (BAU)*	*S2: The strong sustainable development future*	*S3: The intensive regional economic development future*
Forest management	Focusing on improving the overall quality of forest and its resilience to climate change.	Focusing on forest conservation, increasing forest quality and enlarging forest coverage. Higher effort is expected here than in S1.	Maintaining the forests to the status quo level.
International market of agricultural products*	Future prices of beef, milk and coffee will be slightly increased by 10% in the next 20 years.	Future prices of beef, milk and coffee will be largely increased by 30% in the next 20 years (compared with the prices in 2010), and conservation efforts will be more significant, with consequent reduction of conventional agricultural systems.	Future prices of beef, milk and coffee will be largely increased by 30% in the next 20 years, which will dramatically drive the land conversion from forest to agricultural land, and return to conventional pastoral and to conventional pastoral and coffee production to pursue the immediate economic revenues.
Regional sustainable development strategies	Focusing on long-term benefits generated through the promotion of silvopasture and agroforestry.	Focusing on long-term benefits generated through the promotion of silvopasture and agroforestry.	Focusing on maximising the present financial returns of agriculture production rather than long-term benefits of sustainable agriculture.

* In order to understand to what extent international market prices of agricultural products may have an impact on the land-use changes in Costa Rica in general, and in Talamanca in particular, a number of plausible assumptions have been made for the three land-use change scenarios. Since the demand of livestock and other agricultural products has been experiencing a very remarkable increase over the last 50 years in Costa Rica (FAO 2006), it is plausible to assume that this trend may continue in the next 20 years. In particular, in the business-as-usual scenario (S1) a modest 10% increase in the market prices is assumed for beef, milk and coffee in the future as a result of mixed factors, such as the increasing demand in the international market, the expected GDP growth, the increased raw material prices for raising cattle, and the money inflation due to the current economic crisis. Furthermore, to test whether changes of market prices will have different impacts on the strong sustainable scenario (S2) compared with the economic-driven one (S3), an equal increase of market prices for beef, milk and coffee products is assumed in both scenarios, referring to a 30% of increase by 2030. The large increase in market prices of agricultural products is also interpreted as a steering force that drives dramatic land conversions from forests to conventional pastures and coffee fields in Scenario 3, due to the low investment costs and immediate increases in cash incomes to the farmers. Whereas in Scenario 2 (S2), it is assumed that the political choice prefers a strong sustainable development pathway for Talamanca corridor to an economically focused future represented by S3. Therefore, by comparing the socio-economic consequences resulted from the two opposite development paths, the social planner may get a better idea regarding the advantages and disadvantages of future land-use change scenarios in terms of net economic returns to the society as well as to the individual business sectors.

In the present framework, land-use changes are modeled using CA_Markov IDRISI's land-use change model, which includes a multi-criteria evaluation based on different stakeholders' expertise (Clark Labs 2010). First of all, based on the historical land-use trend between 2001 and 2010, a land-use map has been created for a reference year 2010 using satellite images and aerial photographs. The CA_Markov model also contains a multi-temporal land-use change analysis that can create land-use capacity, relief, and ecosystem service maps for the reference year. Next, specific assumptions made for alternative policy scenarios are translated into parameters to recalibrate the CA_Markov IDRISI model and generate new maps of future land uses in VCTBC under three different scenarios by 2030.

Results are shown in Table 6.2 which presents the land coverage for a selection of the most prominent land uses in VCTBC, including forests, pasture land, silvopastoral system, coffee plots and agroforestry systems in 2003, 2010 and projected for 2030 under the three future scenarios.

Table 6.2 Projections of land-use changes under future development scenarios (thousand ha)

Forest	*2001*	*2010 baseline*	*Scenario 1*	*Scenario 2*	*Scenario 3*
Lower montane rainforest	6885.31	6900.08	6900.08	7122.28	6705.25
Lower montane wet forest	1712.86	1712.86	1712.86	2090.29	1690.72
Montane rainforest	1107.86	975.54	924.30	1379.98	1173.37
Premontane rain forest	13,665.38	13,809.31	13,881.18	14,858.61	12,799.46
Premontane wet forest	20,706.62	21,395.05	20,927.61	23,034.61	17,501.93
Premontane moist forest	128.85	134.35	132.15	160.37	133.06
Premontane wet forest transition to basal	395.34	398.37	399.40	445.39	397.08
Very humid forest premountain transition to montane	3092.63	3136.33	3136.33	3312.04	2915.92
Base moist forest, transition to premontane moist forest	434.22	440.14	438.20	515.08	383.76
Tropical wet forest	4042.99	4016.99	4016.99	4082.69	4029.86
Tropical wet forest transition to premontane	5450.88	5690.26	5487.84	6277.18	5493.20
Forest total	57,622.94	58,609.28	57,956.94	63,278.50	53,223.60
Pasture without tree	24,825.28	21,719.17	11,575.72	6765.05	33,861.50
Pastures with tree (silvopastoral system)	4380.93	7239.72	17,429.67	17,029.63	7557.53
Coffee without tree	991.42	973.88	1010.85	1010.85	3008.85
Coffee with shadow (agroforestry system)	8922.74	8764.93	9097.64	9097.64	5097.64
Agriculture land total	39,120.37	38,697.70	39,113.87	33,903.16	49,525.51

The results of the land-use model show that in the BAU scenario (scenario 1), the overall forest cover will be slightly shrunk by 2030 owing to the conversion of some premontane wet forests to coffee plots (including the expansion of coffee area with established agroforestry system) to remain modest growth in coffee revenues. Moreover, some significant land-use changes are found for pasture land. Almost 50 per cent of the traditional pasture land will be converted to silvopasture land, which are more sustainably managed pastures.

In the strong sustainable development scenario (scenario 2), about 5000 ha agricultural land will be converted to forest by 2030. Less than one-third of pasture lands will still be managed under traditional farming practices by 2030. Half of the traditional pastures will be converted to silvopasture and the rest will be converted to coffee plots to meet the increasing coffee demands in the international market.

Finally, the intensive economic development scenario (scenario 3) shows a trend opposite to scenarios 2, where large areas of forests and croplands are converted to pastures for cattle feeding, which are likely driven by the large increase in beef prices in the international market. Moreover, large numbers of farmers decide to drop their agroforestry practices and return to the traditional coffee production in order to pursue higher short-term cash income and increase the farmers' revenues due to the higher productivity and lower operational costs associated with traditional pastures. These changes in land uses may not only have an impact on ES that contribute directly to local stakeholders' income, but also may lead to changes in some ES that are unpriced but vital to support the local livelihoods.

Estimating costs and benefits associated with the provision of ES

All biophysical changes in the flows of different ES are first identified and quantified under the selected future scenarios, and then translated into economic benefits that people perceive.

As discussed earlier, marginal productivity value (MPV) per hectare of land for the annual provision of different ES is required for estimating the total economic benefits generated by different landscapes per year. Different valuation methods can be used to assess these MPV. In the present study, some of the value estimates are taken directly from the economic valuation exercises conducted in previous chapters (see Table 6.10 in the Annex at end of this chapter for the source of information on land productivity for the selected land uses).

For simplicity reasons, we assume a linear relationship between changes in land-use area and time. Therefore, the annual benefit of one type of land use is simply the product of the MPV of the ES and the land coverage of the year under consideration. Moreover, a total of 25 years of land-use change is considered to be consistent with the existing PES program that promotes SPS, as shown in Chagoya (2004), according to which the benefits of timber production can only be realized at the final turn of 25 years, when the timber species reaches its maturity for harvesting.

Forest ecosystems

The MPV per hectare of forest ecosystems is first estimated by forest type and ES under consideration, and then aggregated to obtain the combined economic value provided by forested lands in VCTBC. In the present study, different economic valuation tools are applied depending on the nature of the ES. For instance, a meta-analysis of non-market valuation studies is used to derive the marginal value of recreational services provided by tropical forests (estimated in Chapter 5), whereas a production function derived from a market analysis of revenues is used to estimate the marginal value of *in-situ* water supply for hydropower (estimated in Chapter 4). The latter takes into account the revenues generated in six hydropower plants located near the Talamanca's watershed and analyses its correlation with forest size and runoff. As regards the intrinsic value of biodiversity (non-use value), it is derived from Carranza *et al.* (1996), who used Contingent Valuation Method to estimate biodiversity value in primary tropical forests. In the present study, the value reported in the original study has been converted to the current dollar value, adjusted by inflation and purchasing power parity.

Carbon regulating services provided by tropical forests constitute an essential value of forest ecosystems, which however are not yet formally tradable in the market place. To estimate the total value of forest carbon regulation services, carbon prices in the Emissions Trading System (ETS)[3] is used. By multiplying the carbon price by the annual net carbon flux[4] (which is simply the difference of carbon stocks[5] for a given forest type in two successive years), we can obtain the total value of annual carbon flows provided by all forest ecosystems in the VCTBC area. In this study, a carbon price of $20/tC is used in the valuation exercise, which is approximately equivalent to the average of the highest carbon price of 30€/tC appeared in the European ETS in 2008 and the lowest price of 2.8€/tC in 2013. However, it is important to note that this carbon price refers to a market value, which may not fully incorporate the social damage costs of climate change in the analysis. For this reason, it is suggested that the carbon value presented in this chapter shall refer to a lower-bound estimate and should be refined in the future using the social costs of carbon.

Finally, the market benefits of ecotourism business are estimated based on two value components: (i) the revenues generated from the annual entrance fees to two national parks (Guayaboo National Monument and La Martha Biological Reserve), and (ii) the actual travel costs for tourists to access to the park facilities, including car, tent rental and bus tickets. The historical data regarding the number of visits and prices of entrance fees, car rents and costs of buses are taken from Cifuentes *et al.* (1999) and MINAET (2010). Due to the lack of detailed statistics about the total national/international tourist arrivals by transport mode, as well as the total over-night stays and average spending, the sample has been split equally into two groups by assuming that 50 per cent of the total tourists reach the park by bus and the other 50 per cent by car. This over-simplistic assumption requires better information to be collected in the future to improve the estimates of total revenues that local communities received from eco-tourism related business. The marginal benefits of ecotourism produced on a per hectare basis ($/ha/yr) is estimated by dividing the total tourism revenues of the year by the total forest area.

Table 6.3 presents the final economic estimates of marginal values per hectare of forest ecosystem services, by forest type and under the selected land-use change scenario. These marginal benefits are used to compute the yearly total economic benefits derived from market and non-market ecosystem services between 2005 and 2030. It should be noted that these values represent only the lower-bound estimates, as many ecosystem benefits are unpriced and therefore cannot be captured by economic tools. Table 6.10 in the Annex shows the source of information for each ES.

As for the costs, we consider mainly the costs of forest conservation. In this regard, payments for ecosystem services (PES) estimated by Ecomarket project (financed by the World Bank 2007) is used as a proxy of the total conservation costs, which consist of opportunity costs (OC),[6] management costs (MC), and transaction costs (TC) for the participants. PES is used to compensate the opportunity losses of local farmers due to their participation in the conservation program, and often considered as supplementary revenue to farm income. The underlying reason of using PES to approximate the conservation costs is because in principle, an optimum PES should be set at the level that is exactly equal to the costs of environmental compliance borne by the participants, referring to a Pareto-efficiency, where the redistribution of social wealth from the beneficiaries of ecosystem services to the service providers is balanced through the government mechanism.

Agricultural ecosystems

The total benefits of agricultural ecosystems are mostly derived from the agricultural products sold in the market and vary depending on the land productivity. Many studies in Costa Rica and other Central America countries have shown that planting tree species on pastures (silvopastoral system) and croplands (agroforestry system) can bring about significant multi-benefits to the local communities,

Table 6.3 Estimated marginal flow benefits generated from different forest ES under the selected future scenarios ($/ha/yr)

Forest ecosystem services	*Forest type*	*S1*	*S2*	*S3*
Recreational use of forest[1]	Dry forest	0.93	1.04	0.86
	Moist forest	0.31	0.33	0.31
	Wet forest	5.01	5.23	4.53
	Rain forest	4.66	4.95	4.16
	Forest average	2.73	2.89	2.47
In-situ water supply for hydropower[2]	All watershed	378.76	398.55	346.94
Biodiversity benefits[3]	Forest average	228.23	228.23	76.08
Carbon regulation value[4]	Forest average	7,129	8,051	7,129
Ecotourism revenues[5]	Forest average	7.91	9.49	6.33

Sources: 1. Chapter 5, this volume; 2. Chapter 4, this volume; 3. Carranza *et al.* (1996); 4. Own estimates (Chapter 6, this volume.); 5. Own estimates (Chapter 6, this volume.).

including the increased productivity of beef and milk, enriched level of species on the farmland, increased landscape connectivity, and enhanced carbon stocks, among others. As shown in the future land-use scenarios in VCTBC, changes in land occupation of silvopastoral and agroforestry systems play an essential role in determining whether the regional development is under a sustainable path.

Under the silvopastoral and agroforestry systems, additional revenues can be generated from harvesting timber products 25 years after the system has been established, since the farmers could choose to plant timber tree species with high commercial values on their traditional pastures and coffee plots. Moreover, improved pasture and croplands can also enhance the long-term biodiversity benefits due to the improved connectivity between landscapes, and increase carbon stocks in the growing tree biomass.

In Table 6.4 we summarize estimated annual productivity values per hectare generated by farmland ($/ha/yr) for different agricultural products, as well as the

Table 6.4 Annual economic benefits generated on different farmlands ($/ha/yr)

Agricultural products and forest ecosystem services	*System type*	*S1*	*S2*	*S3*
Milk[1]	Pasture without tree	125	148	148
Beef		3946	4663	4663
Carbon sequestration		2061	2061	2061
Milk	Pasture with tree (silvopastoral)	282	333	333
Beef		7020	8296	8296
Carbon sequestration		3224	3224	3224
Timber products		2806	4,404	1673
Biodiversity benefits due to improved connectivity		228.2	228.2	76.1
Coffee	Coffee without tree	862,016	1,018,746	1,018,746
Carbon sequestration		1890	1890	1890
Coffee	Agroforestry system	985,427	1,164,596	1,164,596
Carbon sequestration		2432	2432	2432
Timber products[2]		2806	4404	1673
Biodiversity benefits[3] due to improved connectivity		228.2	228.2	76.1

Notes: In scenario 3, since the economic future will result in a significant reduction of forest and silvopastoral land, it is assumed that this will negatively affect biodiversity and reduce its value by 50% compared with the baseline in 2010.

Sources: 1. Market prices for milk, beef and coffee in 2010 are collected by CATIE. Original prices measured in $/kg/yr are converted to $/ha/yr; 2. Timber revenues are estimated for trees under natural regeneration in native grassland, taken from Chagoya (2004). It is important to note that the timber benefits are one-time cash income, which can be only realized by the end of the 25 years (in 2030), when all the timber trees reach their maturity and can be harvested together. So, in the balance sheet, this benefit will occur only in the terminal year of the analysis, while the costs of establishing the system are covered by a PES contract, distributed across the initial 3 years (i.e. year 2005, 2006 and 2007); 3. Biodiversity benefits are the non-market benefits estimated from Carranza *et al.* (1996).

additional benefits due to the on-farm tree plantations. Marginal benefits of some of the market and non-market forest ecosystem services are used to assess the additional values derived from these new tree plantations, including carbon sequestration, timber products, and enhanced biodiversity benefits due to improved connectivity.

The total economic benefits per annum generated from various agricultural systems are an aggregation of all yearly ecosystem service values by scenario. These estimated annual marginal benefits generated by farmland are used to compute the total annual economic benefits derived from market and non-market ecosystem services between 2005 and 2030.

As for the costs, Table 6.5 presents all the cost items included in the present analysis. The actual costs per item vary depending on the chosen land-use scenario.

Results and sensitivity analysis

The NPV is calculated at a discount rate of 3 per cent (which is assumed to be equal to the interest rate) for each of the three land-use scenarios during the period between 2005 and 2030, taking into account all benefits and costs in association with different land uses. By comparing the NPV obtained in a strong sustainability scenario (scenarios 2) and an intensive economic future scenario (scenario 3) with respect to the business-as-usual scenario (scenario 1), we are able to conclude

Table 6.5 A summary of all costs associated with agricultural productions

Cost item	*Examples*
Annual maintenance costs for livestock and grassland components	Livestock (from calf to young bull) Degraded grassland Natural grassland Improved grassland
Cost for improved pastures – silvopastoral system (global environmental funds)	Establishment cost Maintenance costs
Costs of maintaining traditional coffee production	Implementation cost of coffee production per ha (initial investments) Maintenance cost of coffee production for the first year Maintenance cost of coffee production for the second year Maintenance cost of coffee production for the third year Management costs Costs of inputs/chemicals in the conventional coffee production
Costs for converting to agroforestry system (i.e. PES for agroforestry contract)	Establishment costs of agroforestry systems Average level personnel costs to work in the plantation + inputs + management costs + others Costs of inputs in the organic coffee production

which future scenario can generate the highest long-term socio-economic benefits. Figure 6.1 shows the comparative results. For instance, it shows that the overall NPV in scenario 2 is 17.5 per cent higher than that in scenario 1 (with approximately 13 per cent increase in the NPV for forest conservation activities and 18 per cent increase in the NPV for improved agricultural land management). On the contrary, choosing the future pathway towards scenario 3 instead of scenario 2 would result in a 6 per cent reduction of forest value due to a 4.5 per cent deforestation of natural forest, which entails a reduction of 6 per cent of total carbon stocked in this land use. All in all, these losses correspond to an economic loss of over 400 million US$ by 2030 in VCTBC.

In order to test the robustness of the results, a sensitivity analysis has been conducted by applying different social discount rates, ranging between 1 per cent and 10 per cent to show different social preferences of consumption. The social discount rate is a reflection of a society's relative valuation on today's wellbeing versus wellbeing in the future. For instance, a low social discount rate indicates that the society values the future's wellbeing derived from long-term economic benefits of ecosystem services as much as today, whereas high discount rate indicates that the social preference for consumption in the society is rather short-term (i.e. it prefers to enjoy the highest wellbeing rather now than later). The results of sensitivity analysis for the three scenarios are summarized in Table 6.6, 6.7, and 6.8, respectively.

The sensitivity analysis shows that although scenario 3 can deliver the highest net benefits of traditional beef, milk and coffee production, these benefits tend to be highly sensitive to the choice of discount rate. As shown in Tables 6.6–6.8, the NPVs of traditional pasture and coffee plots drop more rapidly with the increase of discount rate in scenario 3. The overall NPV estimates reveal that, when 1 per cent discount rate is applied, scenario 1 is preferred over scenario 3, which reflects the fact that the society as a whole gives equal weight to today and future consumption of ecosystem services provided by different land uses. Whereas when 10 per cent discount rate is applied, scenario 3 is ranked higher compared to

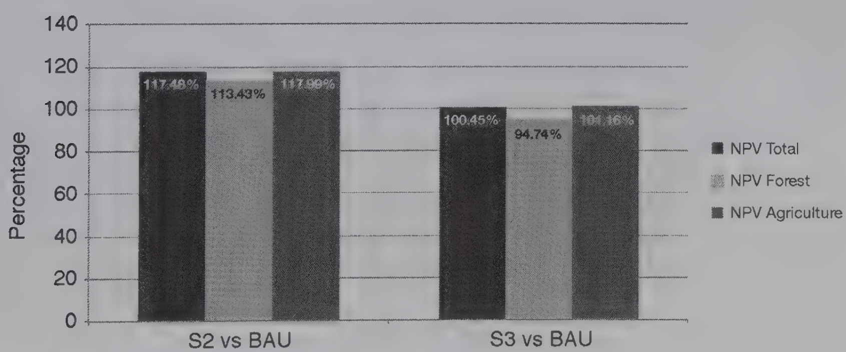

Figure 6.1 Comparison of scenarios 2 and 3 with the BAU scenario (scenario 1)

Table 6.6 SCBA result for Scenario 1: a modest sustainable development future (lifetime: 25 years)

Discount rate (%)	*Overall NPV*	*NPV forest conservation*	*NPV traditional pasture*	*NPV silvopastoral*	*NPV traditional coffee*	*NPV agro-coffee*
1	93,259,253,719	9,868,681,958	485,400,119	879,175,885	5,038,706,965	76,987,288,793
2	80,401,083,048	8,713,059,508	434,795,524	741,617,909	4,444,181,308	66,067,428,800
3	**69,688,202,574**	**7,743,052,596**	**391,791,652**	**628,686,828**	**3,945,574,337**	**56,979,097,161**
4	60,717,745,162	6,924,212,158	355,038,132	535,536,897	3,525,033,926	49,377,924,049
5	53,169,042,253	6,229,084,755	323,450,525	458,344,535	3,168,340,071	42,989,822,367
6	46,785,764,157	5,635,696,310	296,153,983	394,080,594	2,864,116,595	37,595,716,676
7	41,362,142,223	5,126,381,843	272,439,541	340,335,719	2,603,222,919	33,019,762,202
8	36,732,299,376	4,686,878,808	251,730,070	295,185,922	2,378,282,964	29,120,221,612
9	32,761,956,448	4,305,621,944	233,553,662	257,088,734	2,183,318,893	25,782,373,215
10	29,341,961,039	3,973,192,798	217,522,727	224,802,698	2,013,465,267	22,912,977,549

Table 6.7 SCBA result for Scenario 2: a strong sustainable development future (lifetime: 25 years)

Discount rate (%)	*Overall NPV*	*NPVforest conservation*	*NPV traditional pasture*	*NPV silvopastoral*	*NPV traditional coffee*	*NPVagro-coffee*
1	109,622,755,606	11,237,228,844	491,370,215	980,914,555	5,957,531,103	90,955,710,889
2	94,493,924,079	9,913,580,554	443,945,883	827,423,129	5,254,597,230	78,054,377,283
3	**81,890,016,510**	**8,803,259,491**	**403,266,158**	**701,447,987**	**4,665,075,147**	**67,316,967,727**
4	71,336,792,569	7,866,586,941	368,179,689	597,565,593	4,167,856,420	58,336,603,926
5	62,456,784,688	7,071,953,945	337,755,091	511,498,532	3,746,127,360	50,789,449,759
6	54,948,269,744	6,394,070,308	311,235,439	439,859,702	3,386,436,892	44,416,667,403
7	48,569,047,205	5,812,613,504	288,002,766	379,956,242	3,077,977,352	39,010,497,341
8	43,123,877,319	5,311,182,173	267,550,262	329,638,693	2,812,027,444	34,403,478,747
9	38,454,716,713	4,876,482,436	249,460,413	287,184,529	2,581,519,152	30,460,070,182
10	34,433,099,895	4,497,692,852	233,387,738	251,207,911	2,380,699,761	27,070,111,633

Table 6.8 SCBA result for Scenario 3: an intensive economic development future (lifetime: 25 years)

Discount rate (%)	*Overall NPV*	*NPVforest conservation*	*NPV traditional pasture*	*NPV silvopastoral*	*NPV traditional coffee*	*NPVagro-coffee*
1	92,510,201,525	9,315,437,137	801,670,043	559,043,128	10,540,489,455	71,293,561,762
2	80,263,030,355	8,240,105,632	700,477,114	479,497,794	9,043,422,874	61,799,526,940
3	**69,998,557,079**	**7,336,048,705**	**616,236,376**	**413,255,999**	**7,810,528,508**	**53,822,487,492**
4	61,352,471,625	6,571,651,699	545,718,675	357,822,990	6,790,009,914	47,087,268,347
5	54,033,620,077	5,921,695,762	486,362,164	311,212,102	5,940,972,358	41,373,377,691
6	47,808,298,851	5,365,975,405	436,125,851	271,834,541	5,230,999,802	36,503,363,254
7	42,488,105,406	4,888,230,882	393,376,876	238,414,330	4,634,295,194	32,333,788,123
8	37,920,503,466	4,475,320,500	356,803,440	209,922,413	4,130,247,949	28,748,209,164
9	33,981,467,922	4,116,576,481	325,347,276	185,525,408	3,702,326,506	25,651,692,251
10	30,569,728,951	3,803,301,765	298,151,103	164,545,604	3,337,219,166	22,966,511,313

scenario 1, suggesting that people who favor intensive economic development path also prefer to receive immediately high cash income from their land rather than in a distant future.

In contrast, scenario 2 can deliver the highest social benefits, not only because it can deliver the highest benefits for forest conservation, but also because it can balance different land uses to deliver a sustainable sound future. The NPVs are proven to be robust after performing a sensitivity analysis by increasing discount rate from 1 to 10 per cent. In comparison to scenario 3, this result indicates that well-managed silvopastoral and agroforestry systems can generate a NPV that is high enough to compensate the farmers for their economic losses due to the conversion of traditional cropland or pasture to more sustainable systems. This may provide important policy insights regarding how to reallocate resources in the society to achieve sustainable land-use management objectives in a cost-efficient fashion (i.e. to enhance the welfare to all local stakeholders without sacrificing any individual economic benefits).

To conclude, these results suggest that from a social planer's perspective, scenario 2 is the most favorable choice for long-term sustainable development of the VCTBC. From a welfare economic point of view, since the land-use changes mainly occur between traditional pasture to silvopasture, it suggests that resources may be reallocated from the society to traditional pasture farmers for establishing new silvopasture systems on their existing pasture lands. Once the new system is established and well managed, farmers can benefit from long-term economic benefits due to the improved land-quality (e.g. carbon credits), increased agricultural productivities (i.e. higher productivity of milk, beef and coffee, due to increased shaded area) and additional sources of incomes (e.g. timber revenues from the planted trees). In addition, keeping aside some lands for silvopastoral and agroforestry systems may also safeguard the stability of farm incomes during a bad year with an unfavorable climate condition, as prices for silvopastural and agroforestry products are relatively stable all year round in a niche bio-market compared to the conventional market. Notwithstanding the agro-tourism is not tackled in the present study, it is obvious that improved pasture land and coffee fields will have great potential in terms of developing new business as such. From this viewpoint, the net economic benefits of silvopastoral and agroforestry systems are underestimated, as some benefits have not been included due to the lack of data. It is, however, worth of noting that the PES scheme has played an essential role at the initial stage in helping the farmers to establish the system.

Conclusions and policy implications

This chapter proposes a methodological framework to assess costs and benefits associated with changes in land-use types and to evaluate trade-offs and synergies of conservation strategies for biodiversity conservation and provision of ecosystem services. The model builds on a SCBA, which incorporates physical indicators on land-use changes projected under future scenarios and taking into account policy and economically driven factors.

The proposed approach is designed to assist local policy-making to set policy priorities and identify land-use management options providing the highest co-benefits, while incorporating the interests of different stakeholders. Notwithstanding the approach has been applied to a specific location in this study, the SCBA framework is also applicable to other contexts where a decision has to be taken between conflicting land-use management strategies.

The results obtained from the SCBA exercise are very promising, showing interesting insights for policy-making to improve the land-use management in the VCTBC (see Box 6.2). Not surprisingly, results from NPV estimation suggest that, among all others, scenario 2 (the strong sustainable development future) delivers the highest social benefits.

The results are very robust with increasing the discount rate, indicating that from a social planer's perspective, scenario 2 is the most favorable choice for the long-term sustainable development of the VCTBC, as it can balance different land uses to deliver a sustainable sound future. In particular, it shows that sustainable farming practices, such as agroforestry and silvopasture with the objectives of increasing the connectivity between fragmented farmlands and enhancing biodiversity do help to improve the ecosystem services and deliver higher social benefits. This reaffirms the previous research findings reported by the InVEST project (Integrated Valuation of Ecosystem Services and Tradeoffs) that improving biodiversity conservation also enhances ecosystem services, indicating a synergy between the two (Nelson *et al.* 2009). However, it does not necessarily mean that land management practices that focus on enhancing ecosystem services can also provide good outcomes for biodiversity, which has recently been demonstrated by Macfadyen *et al.* (2012).

Moreover, our results suggest that well managed silvopastoral system and agroforestry system can generate a NPV that is high enough to compensate the farmers for their economic losses due to the conversion of traditional cropland or pasture to more sustainable silvopastoral and agroforestry systems. However, it must be said that local developing world communities often bear the initial opportunity costs of conservation. Inversely, the greatest benefits, which include the sustainable provision of ES (food, timber, other fibers, medicines, nature-based tourism, nutrient and climate regulation, carbon storage, option, existence and bequest values, the latter being associated with the conservation of the ecosystem for future use or appreciation), are often widely dispersed among wealthier national and global beneficiaries (Balmford and Whitten 2003). In cases of such dramatic socioeconomic contrast between those who receive the greatest benefits, versus those who bear the greatest costs of ecosystem service protection, governments will have to play a central role in creating the necessary international payment mechanisms (Balmford and Whitten 2003), such as international payment for ecosystem services (IPES), fair trade, labeling, bans, international biodiversity offsets in private sector and PES-based landscape labeling (Levine and Chan 2011). In the case of VCTBC, it is worth of noting that the PES scheme has indeed played an essential role at the initial stage to cover the initial opportunity costs and help farmers to establish the system.

Box 6.2 Key policy messages delivered by this research

- Farming practices that increase or protect biodiversity in an agricultural landscape will often indirectly help preserving ecosystem services, but actions that focus on enhancing ecosystem services will not necessarily provide good outcomes for biodiversity. In other words, synergies between agricultural productivity and biodiversity conservation can only be achieved if an understanding of ecosystem services leads to a change in management practices that supports greater biodiversity.
- Well-managed silvopastoral and agroforestry systems can generate a NPV that is high enough to compensate the farmers for their economic losses due to the conversion of traditional cropland or pasture to more sustainable systems. However, the initial opportunity costs may be too high, preventing many farmers from participating to the program. Therefore, governments will have to play a central role in creating the necessary international payment mechanisms (Balmford and Whitten 2003) to support local conservation activities, provided that the greatest benefits are often widely dispersed among wealthier national and global beneficiaries. Existing financing instruments include Payments for Ecosystem Services (PES), Green Infrastructure, and Biodiversity offsets.
- In addition, national/local authorities may also create financial incentives by promoting new/innovative business models related to biodiversity conservation, such as the pro-biodiversity business/certified products, eco-tourism, habitat banking, and bio-carbon markets, financial services, which can kill two birds with one stone (i.e. it can not only help generate revenues of small and medium enterprises, but also can increase the positive ecological impacts and mitigate/reduce the negative impacts of business operations on biodiversity).

Finally, the results also show that financial cash incomes generated from land-uses are sensitive to the changes of discount rate, revealing the fact that the choice of future development pathways is highly affected by the way the society would weigh future consumption of ecosystem services. If people weigh higher their present cash income from the land use, then scenario 3 is the second best development path for Talamanca. On the contrary, if people give equal weight to guarantee their future consumption of the same amount of high quality good as today, then scenario 1 offers the second best development path. However, in the developing communities, although people may be able to recognize the importance of a sustainable future, it is very likely that they cannot bear the financial losses due to the declined land productivity. Therefore, to ensure the regional

development moving under a more sustainable pathway, the local governments need to develop innovative economic instruments to ensure the availability of sufficient funds that can be used to compensate farmers' financial losses as a result of participating in the sustainable farming programs. In addition, they may also create incentives by promoting new business models, such as pre-biodiversity certified products, eco-tourism, habitat banking, bio-carbon market, etc. related to biodiversity conservation.

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Annex

Table 6.9 Scenario description

Scenario	*Description*
1. The modest sustainable development future	In this scenario, the policy emphasis is to maintain the current status of natural forests, and to prevent forest degradation from climate change. In addition, prices of beef and coffee products in the international markets are expected to slightly increase over the next 20 years. As a result, the expected land value will be relatively stable by 2030, which implies that the opportunity costs of forest conservation will remain low. For this reason, the present analysis does not consider future trends in land value change over time in the Talamanca corridor.
2. The strong sustainable development future	In this scenario, conservation of forest and biodiversity is the focal area of the policy making. The corridor management will take actions in terms of increasing the coverage of secondary plantations (which will grow to natural forests over a long period of time) and enlarging the areas of silvopastoral system and agroforestry system in Talamanca by increasing the tree density on the improved natural pastures or coffee farms. The prices of agricultural products are expected to increase significantly in the international market, reflecting a higher value of agricultural land, and thus higher opportunities costs of forest conservation. This is however neglected in the current cost analysis due to a lack of sufficient market information, but it shall be addressed and considered in future research. Under this scenario, it is therefore reasonable to assume that the protected forest area will be enlarged as a result of the effective measures under Sustainable Forest Management in the corridor. Moreover, plantation of timber species on pastures and cropland will be also encouraged and implemented in Talamanca. This action will generate higher incomes to farmers through additional timber products and increased farm productivity. However, high investment in establishing and maintaining the new farming systems shall be financed by new economic mechanisms, such as the PES scheme.
3. The intensive regional economic development future	This scenario assumes a situation opposed to Scenario 2. In this scenario, the land uses and management practices will focus on how to generate higher incomes to the local farmers, from their engagement in agricultural activities. Pursuing the greatest financial return from the lands is the key development strategy, rather than balancing the forest conservation benefit and agricultural revenues. Prices of beef and coffee in the international market will significantly increase, driven by the increasing demand for beef and coffee abroad. Although the current land-use and management policies in Costa Rica favour forest protection and natural conservation, there is no guarantee that the measures will not be affected by dramatic changes in market prices of key agricultural products (Calvo-Alvarado *et al.* 2009). Thus, under this scenario, there is a high probability that extremely high net returns from agricultural products may trigger new trends of conversions from forests to agricultural lands.

Table 6.10 Sources of data on land productivity for forest, pasture, silvopasture, coffee, and agroforestry coffee plots

Ecosystem/ agro-systems	*Benefits*	*Source of info on land productivity ($/ha)*
Tropical forest	Hydrological services including provision of water, for human consumption, irrigation, and energy production	Chapter 4 projections
	Recreation	Chapter 5 projections
	Ecotourism	CATIE field data (original data set is available on request) and MINAET (2010)
	Carbon sequestration	Own estimates (Chapter 6)
	Biodiversity benefits	Harvey *et al.* (2008)
Pasture without tree	Milk	Casasola *et al.* (2009)
	Beef	CATIE field data (original data set is available on request)
	Carbon sequestration	Own estimate (Chapter 6)
Pasture with tree (silvopastoral)	Milk	Casasola *et al.* (2009)
	Beef	CATIE field data (original data set is available on request)
	Carbon sequestration	Own estimate (Chapter 6)
	Timber production	Chagoya (2004)
	Biodiversity benefits due to improved connectivity	Own estimate (Chapter 6)
Coffee without tree	Coffee	CATIE field data (original data set is available on request)
	Carbon sequestration	Mena (2008)
Coffee with tree (agroforestry)	Coffee	CATIE field data (original data set is available on request)
	Carbon sequestration	Own estimate (Chapter 6)
	Timber production	Chagoya (2004)
	Biodiversity benefits due to improved connectivity	Own estimate (Chapter 6)

Notes

1 SAPs were introduced by the World Bank to decrease the profitability of agriculture and cattle ranching in marginal forestlands.

2 Costa Rican policies have established incentives to create special conservation areas, and promote reforestation and forest management, including income Tax Deduction (1979), Soft Credits (1983), Forest Payment Title (Certificado de Abono Forestal, CAFs, 1986), Fund for Municipalities and Organizations (1986), Forest Advance Payment Titles (Certificados de Abono Forestal por Adelantado, CAFA 1998), and Fund for Forest Development (1998) (De Camino 2000).

3 The ETS was launched in 2005 as a result of the European Union's policy to combat climate change and its key tool for reducing industrial greenhouse gas emissions cost-effectively. The ETS works on the "cap and trade" principle, under which a cap is set on the total amount of greenhouse gases that can be emitted by all participating installations. "Allowances" for emissions are then auctioned off or allocated for free, and can subsequently be traded. Installations must monitor and report their CO_2 emissions, ensuring they hand in enough allowances to the authorities to cover their emissions. If emission exceeds what is permitted by its allowances, an installation must purchase allowances from others. Conversely, if an installation has performed well at reducing its emissions, it can sell its leftover credits. This allows the system to find the most cost-effective ways of reducing emissions without significant government intervention.

4 This method is adapted from the so-called "stock change method," which is developed by the IPCC good practice guidance for land use, land-use change, and forestry (Penman *et al*. 2003).

5 Total carbon stock by forest type is a product of carbon density by forest type and the forest area. The carbon densities (tC/ha) of forest land in the Talamanca area can be found in Cifuentes-Jara (2008).

6 Opportunity costs of forest conservation in Talamanca corridor refer to the forgone benefits from other land uses, such as agriculture and real estate.

Part III

Economic assessment, adaptation options and policy implications

7 The role of economic valuation of ecosystem services in an interdisciplinary context

Anil Markandya

Introduction

The placing of a monetary value on the services provided by ecosystems has become a major activity of environmental economists. To be sure, there are detractors, who claim that the dollars or euros attached to some services undervalue the natural systems, which have a range of functions that are not capable of being valued in these terms. But there is also growing evidence that even if we take only the services that have been valued the numbers are very large and can tilt the balance of decision-making when evaluating development or conservation options in favor of approaches that protect the natural resource base.[1] Some authors (Adamowicz 2004; Silva and Pagiola 2003; Smith 2000) have looked at the use of valuation of ecosystems for policy purposes. They find that some actions have indeed resulted from the work that has been done in this area, such as damage assessment cases in the USA, controls on some pollutants based on evaluations of human health, cost–benefit analysis of water resource planning, forest resource use planning, and tax revenues from the improvement of the environmental quality. However, there have not been as many applications as one would hope for, and the majority of the ecosystem valuation studies have been of an academic nature and have not intended to influence decisions (Markandya and Pascual 2014).

To understand the role of economic valuation we have to see it in the context of the problem that is being addressed. This book mainly looks at selected functioning ecosystems in Central America and the services they provide now and will provide in the future under different climate scenarios. It discusses the most suitable biophysical models for each of the ecosystem services that are studied. For hydropower the authors used the MAPSS model, while the water and recreational services are analyzed with the use of the Holdridge zone model, in which four "life zones" or "Holdridge systems" are identified, each characterized by bio-temperature (temperature range allowing vegetative growth), annual precipitation and humidity. In each case biophysical models aim to provide a link between the state of the system and the quantity and quality of the service provided. With climate change the functioning and size of the biophysical systems will alter and as a consequence so will the values derived from them.

An important contribution of this book is the central role given to spatial analysis. Each of the ecosystems analyzed has been done in a spatial context, which serves two important functions. First the estimates of the impacts are more accurate than they would be from a more aggregated approach and second the results are more useful for policy-makers because they provide a breakdown of where the gains and losses occur and who benefits and who loses.

The rest of this book goes into the valuation questions in great detail and it is not necessary for me to discuss them in this chapter. Rather I am going to discuss how we can use these estimates for policy purposes. There are many questions to be addressed in linking the monetary values obtained to the decisions that have to be taken. Perhaps a good point of departure is to set out what the long-term future for the region will look like in economic and social terms. This is done in the next section. I then ask how the expected changes in ecosystem services will impact on values associated with them under these broad future scenarios and what measures can be taken to adapt the systems in the short to medium term so as to get the best use of them over the rest of this century. Some of these adaptation questions are also addressed elsewhere but the perspective offered here is a wider one and also comments on the cost–benefit analysis and adaptation proposals of Chapters 6 and 8.

The development and climate scenarios to 2100

What kind of future will Central America face under different climate scenarios? To begin with let us look at the socio-economic structure now. We are dealing with a region made up of seven countries and a population of around 42 million. They vary a lot in terms of living standards: from a low per capita income of $2,100 in Nicaragua to a high of nearly $11,000 in Costa Rica. In terms of urbanization they also vary: from around 47 per cent in Guatemala and Honduras to around 60 per cent in Costa Rica and El Salvador, and 71 per cent in Panama.

In the most recent climate literature the future is analyzed through a set of socio-economic pathways (SSP),[2] which combine climate and development aspects to create a set of scenarios or pathways. There are five such pathways currently under consideration, which can be summarized as follows:[3]

- *Scenario 1:* Sustainability scenario, where we see declines in high fertility areas, there is convergence in per capita incomes and there is fast pace of low carbon technology development.
- *Scenario 2:* A middle of the road scenario, where there is less decline in fertility, slower growth in per capita income and a slower pace of technology development.
- *Scenario 3:* A fragmentation scenario, with continued high population growth in low-income countries, slow growth in per capita incomes, and low pace of low-carbon technology development.
- *Scenario 4:* An inequality scenario, where the developed world grows faster than the developing world and technology developments are not shared through transfer.

- *Scenario 5*: Conventional development scenario, where current trends of fossil fuel use continue with little attempt to shift to a low carbon trajectory. Growth trends continue as at present.

In Table 7.1 the implications of these scenarios for the countries of Central America are given. Some things stand out in the futures that are shown there. First the living standards the citizens of these countries will enjoy depend critically on which scenario is realized. Take Belize, for example. If the sustainability scenario or the conventional development scenarios hold, GDP per head will go up from around US$8,000 in 2010 to US$58,000 or US$92,000 in 2100. But if a fragmentation or inequality scenario are realized the figures will be US$19,000 or US$20,000 in 2100.

The economic policy-makers in these countries will be presented somewhat as in Table 7.1 as a basis for their medium- to long-term planning. Hence it is important to see how this information base links into the ecosystem assessment that has been carried out by the specialist teams in the other chapters.

Projecting forward

The outstanding fact that emerges from the scenarios is the degree of variation: we face a future with huge uncertainties, all of which are relevant to the values we associate with the ecosystem services that are being valued and with the possible adaptation options. Let us consider each of the areas individually.

Water

Water is used for agriculture, household drinking and other domestic needs, and energy generation. It also determines the recreational experience by affecting the fauna and flora in the different zones. For agriculture the value of water will depend on the crops chosen to be grown and on the prices of these crops. Choices about crops will depend on the changes in water availability as well as changes in its quality, both influenced by the climate scenarios.

The estimated values of such water services in Chapter 5 of this book apply current estimates of water services per hectare in different watersheds in Mesoamerica to the expected areas under climate change. These estimates have been carefully derived from a review of the literature, which gives values of water services per hectare for different forest zones. Climate change is expected to affect the size of the different types of forest (dry, moist, wet and rain forest) and the magnitude of this impact requires sophisticated modelling of responses of forest types to changes in temperature, rainfall, and so on. As noted elsewhere in the book the runoff is affected in non-linear and complex ways by changes in vegetation.

The project has analyzed these very well using the climate models to provide data on the climatic variables, which depend on the scenarios. But as we have seen above, the same scenarios also imply changes in income and populations and

Table 7.1 Present data and future estimates of GDP and population for Central America

Country	*Scenario*	*GDP Per Capita*				*Population (Millions)*			
		2010	*2020*	*2050*	*2100*	*2010*	*2020*	*2050*	*2100*
Belize	Scenario 1	8345.292	10884.22	22327.28	58449.48	0.311627	0.35592	0.389792	0.294765
	Scenario 2	8345.292	9968.395	17420.5	43444.29	0.311627	0.3648	0.447515	0.44874
	Scenario 3	8345.292	9409.896	12777.35	18897.23	0.311627	0.374871	0.562643	0.849569
	Scenario 4	8345.292	9341.641	12288.85	20076.21	0.311627	0.371892	0.515932	0.740769
	Scenario 5	8345.292	10999.63	25007.42	92132.47	0.311627	0.352642	0.347957	0.227785
Costa Rica	Scenario 1	10736.03	13946.13	23608.95	56377.39	4.658887	5.250641	6.175663	4.593338
	Scenario 2	10736.03	13349.56	19680.36	45289.67	4.658887	5.326162	6.735972	6.455497
	Scenario 3	10736.03	12582.72	13823.52	19195.72	4.658887	5.362279	7.09318	9.199059
	Scenario 4	10736.03	12586.81	13705.93	19309.49	4.658887	5.300536	6.349723	5.047726
	Scenario 5	10736.03	14053.15	27492.14	93644.52	4.658887	5.290639	6.473902	4.975962
Guatemala	Scenario 1	5802.268	8627.695	19548.91	52294.92	14.38893	16.88136	20.61217	18.15721
	Scenario 2	5802.268	7820.45	14669.11	39825.79	14.38893	17.50524	24.96549	28.8932
	Scenario 3	5802.268	7175.248	9275.758	13414.68	14.38893	18.27753	32.85582	57.11629
	Scenario 4	5802.268	7124.764	9313.168	15937.16	14.38893	18.16072	31.3758	53.00766
	Scenario 5	5802.268	8706.65	22672.32	87495.52	14.38893	16.73915	19.14844	15.63664
Honduras	Scenario 1	3718.464	5066.175	13059.08	37316.86	7.600524	8.696787	10.09401	8.239897
	Scenario 2	3718.464	4609.446	8958.991	27628.31	7.600524	8.923785	11.65861	12.12244
	Scenario 3	3718.464	4181.632	5045.042	7021.397	7.600524	9.244708	14.59451	22.00068
	Scenario 4	3718.464	4167.461	5424.185	10164.52	7.600524	9.185156	13.8835	20.30288
	Scenario 5	3718.464	5120.394	15650.16	67183.22	7.600524	8.624685	9.408531	7.161154

Table 7.1 Continued

Country	*Scenario*	*GDP Per Capita*				*Population (Millions)*			
		2010	*2020*	*2050*	*2100*	*2010*	*2020*	*2050*	*2100*
Nicaragua	Scenario 1	2096.05	2850.279	8794.05	30208.6	5.788163	6.234509	6.146741	4.285718
	Scenario 2	2096.05	2660.24	6067.372	22715.63	5.788163	6.36554	7.013668	6.243738
	Scenario 3	2096.05	2490.132	3420.976	5338.597	5.788163	6.639275	9.277361	12.51367
	Scenario 4	2096.05	2512.708	3810.386	8383.304	5.788163	6.332815	6.499464	4.708422
	Scenario 5	2096.05	2886.599	10649.43	53760.34	5.788163	6.112193	5.193042	2.943759
Panama	Scenario 1	9732.77	15508.9	26247.18	53888.73	3.51682	3.964649	4.644976	3.621941
	Scenario 2	9732.77	15159.33	23095.82	43142.15	3.51682	4.021979	5.108167	5.024697
	Scenario 3	9732.77	14808.99	17969.23	24742.21	3.51682	4.068959	5.659287	7.57108
	Scenario 4	9732.77	14780.23	17598.25	24854.17	3.51682	3.986426	4.711122	3.662801
	Scenario 5	9732.77	15646.62	30592.51	92418.22	3.51682	3.970177	4.67022	3.64436
El Salvador	Scenario 1	6175.372	7469.215	18161.39	44474.18	6.192993	6.286917	5.425292	3.318179
	Scenario 2	6175.372	7152.542	13899.01	35676.47	6.192993	6.383133	6.080511	4.757024
	Scenario 3	6175.372	6797.073	9608.092	15402.76	6.192993	6.675849	8.347277	10.48711
	Scenario 4	6175.372	6828.64	9551.44	17907.9	6.192993	6.32194	5.561416	3.5528
	Scenario 5	6175.372	7561.989	21221.32	76043.16	6.192993	6.117551	4.215849	1.733416
Central America	Scenario 1	5864.014	8244.515	18018.24	47174.25	42.45794	47.67078	53.48864	42.51105
	Scenario 2	5864.014	7697.403	13805.5	36371.69	42.45794	48.89064	62.00993	63.94533
	Scenario 3	5864.014	7167.152	8894.831	12769.36	42.45794	50.64347	78.39008	119.7375
	Scenario 4	5864.014	7166.076	9023.277	14915.26	42.45794	49.65948	68.89695	91.02306
	Scenario 5	5864.014	8349.537	21345.62	81575.7	42.45794	47.20704	49.45794	36.32308

Source: SSP Database (https://secure.iiasa.ac.at/web-apps/ene/SspDb/dsd?Action=htmlpage&page=series).

these changes are likely to be significant. Indeed they could even dominate the impacts of changes in forest areas. Moreover, they have a very wide range of possible projected values. Thus the value of water will have a much wider range of benefits than is currently assumed, depending on which scenario is realized.[4]

Recreation

Similar issues arise with regard to the recreational services provided by tropical forests. The meta-analysis picks up the impact of climate change in terms of the changes in land areas under different forest zones (Chapter 5). These area changes in turn affect the values associated with recreation. But so do the changes in real incomes and population and the changes in the latter two can be very large and very varied across the scenarios.[5]

Energy

The hydrological model that estimates water flows (Chapter 3 and 4) provides the basis for the estimation of the changes in hydropower generation capacity. The application is at the watershed level and the results vary considerably by watershed (with those in Costa Rica and Panama most severely affected) and by climate scenario: the greater the expected future emissions the greater the impacts. The value of losses, however, will also depend on future demand for energy and on the willingness of citizens to pay for it. These variations also need to be taken into account when looking at possible future options for action.

Implications for adaptation

How would a government that was concerned about climate change and wanted to take action in the area deal with such information? It would start out by noting a few key features of the problem. First is the long time horizon—the estimated losses are for 2100, a year so far away as to suggest that the problem is not urgent. Second it would note the wide range of possible outcomes, both in terms of the real potential impacts as well as the perceived world that will exist at that date. Given these undisputed facts decision-makers will be tempted to put the problem on the back burner and say that they will take action closer to the time, when they have better information about what is likely to be the outcome.

There is some justification in taking such a position, given the many priorities that need to be addressed in the short term, such as poverty, unemployment and the shortage of resources. Yet the evidence from the climate change literature suggests that to ignore the long-term problems could be a mistake, because there are measures we can take at modest cost now will reduce climate damages in the future significantly (Wang and McCarl 2013).

The ecosystem valuations described above do not directly provide this information but other work in the book goes in that direction. The land-use change model discussed in Chapters 6 examines options over the relatively short

term (to 2030) and estimates their benefits. It considers three alternatives: a strongly sustainable package of measures in terms of forest conservation and agricultural practices, a moderately sustainable one and one that consists of intensive development with little attention to conservation. These are compared against the costs of the packages and the authors find that the strong sustainable development package has the highest net present value. Such an analysis could not have been carried out without detailed valuation of the ecosystem services discussed above. The valuation, however, extends only over 25 years and what is missing is the link between these different scenarios and developments over the long term. It would be interesting to take the same three packages and track them further into the future when climatic effects start to bite. Would the more sustainable package be less affected under climate change? Or, to look at it differently, what additional actions would we need to include in the strongly sustainable package *now* to make the forest systems more resilient in the future under climate change? This should surely be the subject of future research, which can build on the sound work undertaken to date. If, as I expect, we find that the modified strong sustainability package generates higher benefits in the long term as well, then policy-makers can be persuaded to fund it on two grounds: the short-term gains as well as the potential long-term ones.

An issue of some importance in determining which package is the most preferred is the choice of the discount rate. Incurring some additional costs today so as to generate benefits in the distant future will only be justified if the discount rate is low enough. Even over the relatively short period to 2030 a discount rate of 10 per cent turns out to result in the intensive development options being chosen as the best. With a longer-term program to 2100 the use of a rate of close to one per cent will be needed, or alternatively a declining discount rate will be required. Declining rates are applied for projects with long-term gestation periods, so that distant costs and benefits are discounted less than the near-term ones (see HM Treasury 2003 for further details).

The other issue that comes out of ecosystem valuation work in a climate policy context is the exceptionally large uncertainty. The present analysis in the book does not pay as much attention to this as it deserves but in this respect it is no different from a lot of other work on climate adaptation. The project would be enriched if it considered how certain adaptation measures could be robust under different scenarios. If, for example, actions to strengthen the Volcánica Central Talamanca Biological Corridor could be shown to provide significant benefits under the whole range of climatic scenarios, as well as benefitting communities today, then the case for the action would be strengthened. It does, however, need to consider the wide range of climatic outcomes and to build in the socio-economic dimensions.

Conclusions

The valuation of ecosystem services in monetary terms provides a very important input to making the case for the conservation of biomes that provide these

services. The estimates are certainly not precise and they do not cover all important aspects of these ecosystems but they are, nevertheless, useful, as this study shows.

In carrying out a study involving changes to ecosystems it is essential to have sound biophysical modelling that informs us about the changes to the systems as a result of external factors. It is these changes that we seek to value. In doing so we need to recognize many uncertainties, not only in the results of the biophysical modelling, but also in the socio-economic context in which the values are embedded. In some cases these socio-economic uncertainties can dominate the estimated values and climate change is a case in point, as we have seen here.

The information collected on ecosystem services will most likely influence policies for adaptation to climate change if (a) it can be shown that the benefits are present under a wide range of possible futures, (b) the costs today are modest, and (c) there are short-term benefits as well, especially to vulnerable groups. Sometimes these short-term benefits to adaptation are referred to as co-benefits. In addition, if we can show that failure to act now will imply even higher costs in the future, the case will be even stronger.

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Notes

1 Indeed, some estimates have put the values of the world's ecosystems in trillions of dollars. Costanza *et al.* (1997) estimated it to be around US$33 trillion a year, at a time when global GDP was only US$18 trillion. Such a value, however, has a methodological problem. If all such ecosystems disappeared life as we know it would not be possible and the loss would be total. In that sense ecosystems have an "infinite" value

and as someone noted a study such as this provides an underestimate of infinity. The interesting valuations of ecosystem services are those related to changes in the level of the service in a practical context and these can also be large but credible.

2 See https://secure.iiasa.ac.at/web-apps/ene/SspDb/dsd?Action=htmlpage&page=series. Enter as guest. Accessed June 10th 2014.

3 Other chapters also consider climate scenarios, some of which are slightly different. I present these here as the latest and, most importantly, as giving the key economic and demographic projections relevant to planning and adaptation.

4 The meta-analysis indicates that income growth will result in increases in the value of water services while population growth will result in decreases. The former is obvious while the latter is less so; perhaps it indicates that areas with higher population have more degraded systems with lower water services.

5 The meta-analysis function has income and population affecting recreation values in the same way as they affect water values. This may be difficult to justify as the demand for the respective services responds differently to these two variables. Future work may look into that question.

8 Ecosystem-based adaptation

Nature-based responses to climate change impacts

Giacomo Fedele, Raffaele Vignola and Marco Otárola

Introduction

Societies have always sought to protect themselves and their valued assets from natural pressures and reduce their vulnerabilities (e.g. the hunting of large predators, the suppression of wildfires and the location of settlements in strategic areas). In modern times, engineering solutions have been widely used to safeguard infrastructure and productive systems from various hazards. Slopes have been stabilized with terraces, while rivers and coastal areas have been modified with dams and seawalls to regulate floods and provide irrigation. In recent years, increasing interest is being directed towards adaptation approaches that use ecosystem services to build socio-ecological resilience for extreme climatic events. From a socio-ecological perspective, resilience is characterized by the amount of change that a coupled system can undergo and still retain the desired functions and structure (resist); the degree to which it is capable of self-organizing (recover); and its ability to build and increase learning capacity and to adjust (adapt) (Gunderson and Holling 2002; Trosper 2002; Magrin *et al.* 2007). Improving the resilience of both ecosystems and people is one of the most readily available and accessible strategies for responding to unwanted changes and risks such as those caused by climate variability.

Ecosystem-based adaptation (EbA) is defined as a set of adaptation policies or measures that consider the role of ecosystem services to respond to the adverse impacts of climate change and can be used at multiple scales and in different sectors (CBD 2009; Vignola *et al.* 2009). EbA initiatives support development aspirations and adaptation objectives through the sustainable management of biodiversity and ecosystems (Naumann *et al.* 2011). Healthy natural and semi-natural ecosystems provide a range of services for people's well-being (e.g. fuelwood, clean water, raw materials, medicines, shelter and food). In addition, ecosystems form natural buffers against extreme weather events, thus supporting the resilience of people to climate variations and hazards (CBD 2000). For example, revegetating a degraded steep slope with trees helps to reduce the risks of landslides by protecting the soil from erosion. At the same time, trees can increase the food security of the local community by providing fruits and fuelwood.

Although EbA is mostly intended to decrease people's vulnerability, it should also aim to reduce ecosystem vulnerability (e.g. SBSTA 2013). Both socio-ecological systems are intrinsically interconnected and if ecosystems are not able to adapt to climate change, their ability to provide benefits would be compromised with negative consequences on people's vulnerability (Locatelli *et al.* 2008). Similarly, EbA has been described to be "about saving ecosystems and about using them to help people and the resources on which they depend" in the face of climate change (Burgiel and Muir 2010). The use of "green," "soft" or "ecological engineering" as strategies of defense against climate change is particularly relevant considering that "in most places in the world, nature is the single most important input into local economies and human well-being" (Roberts *et al.* 2011). Ecosystems, in contrast to hard engineering measures such as steel or concrete infrastructures, are often immediately available and more accessible and integrated into communities (CBD 2009). EbA has synergies with community-based adaptation approaches and can effectively build on local knowledge and needs, while providing particular consideration to the most vulnerable groups of people, including women and the poorest, and to the most vulnerable ecosystems (see Figure 8.1).

This chapter focuses on EbA in forest ecosystems and the provision of additional benefits whose economic and ecological impacts are discussed along the book. In the first section below we discuss the process needed to implement EbA. The following section illustrates possible EbA applications with three case studies from Central America in which the resilience of socio-ecological systems toward climate change was strengthen. The case studies show how EbA support resilience while generating additional benefits relevant to reduce the negative impact of climate variability on other economic and ecological factors such as carbon stocks and timber, water regulation, and recreation. Finally, we summarize the main points and the policy implications in the Central America context.

Ecosystem-based adaptation implementation

People enhance or maintain particular goods or services of interest by altering the type, magnitude, and relative mix of services in landscapes. These interventions build the technical basis of EbA and include the sustainable management, conservation and restoration of ecosystems (CBD 2009). There is a demand to see concrete examples translated into practice, but these remain limited due to the relative novelty of the EbA concept (World Bank 2010; Pramova *et al.* 2012). So far, most of the evidence for successful EbA interventions is embedded in studies that are not explicitly labeled as such, but focus on the impact of ecosystems on biophysical parameters without linking them directly to socioeconomic benefits or adaptations (Doswald *et al.* 2014). Several other EbA relevant experiences can be classified outside of adaptation-oriented studies, in ecosystem restoration, soil and water conservation, and disaster risk reduction. Lessons learnt in these fields are contributing to develop good practice for EbA (see Box 8.1).

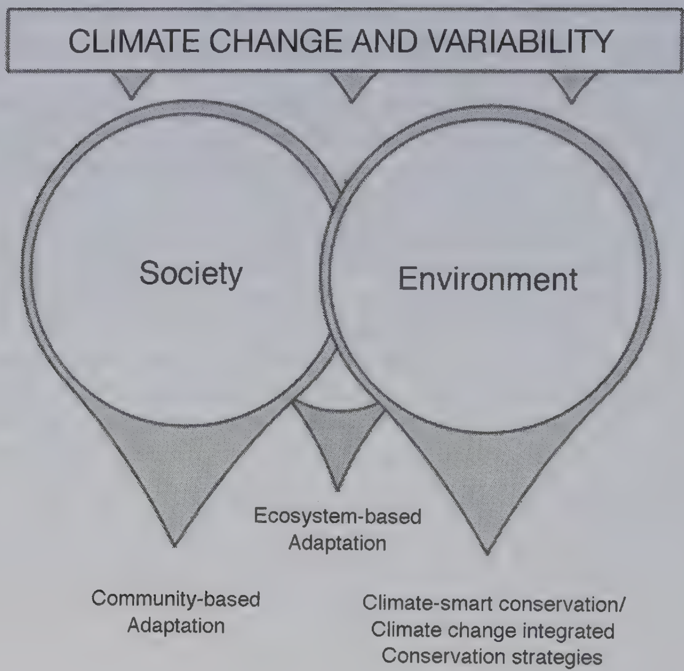

Figure 8.1 Socio-ecological systems and related management strategies for climate change adaptation

The choice of the EbA intervention to be used strongly depends on the context. The consequences of climatic variations can be manifested in different ways according to local characteristics (socio-economic and biophysical). Addressing climate vulnerability must consider a *particular* outcome (e.g. food security) of a *selected* system (e.g. a rural village) to a *given* exposure (e.g. flood) within a *definite* time horizon (e.g. in the last 10 years). Accordingly, generalizations about the climate vulnerability of a place can be misleading (Luers *et al.* 2003; Füssel 2007) and measures cannot be designed in a "one size fits all" manner. In order to develop appropriate EbA interventions that consider the local context and are relevant for different time scales, we require a clear understanding of the characteristics and dynamics of the socio-ecological systems. We need information on (i) climate variation and impacts, (ii) expected changes, (iii) ecosystem services and trade-offs across time, space and social groups, and (iv) progress towards the ideal state. This knowledge can be gained through several studies that constitute crucial steps for good EbA implementation. They usually involve the gathering of information from the laboratory and fieldwork; metadata reviews of literature, modeling and scenario-building; expert opinions and local knowledge from different groups.

Box 8.1 Ecosystem-based adaptation principles

EbA-relevant measures have long been applied by communities in response to the effect of climate variations, sometimes supported by a wide range of organizations from the field of conservation, development and disaster reduction. These experiences have led to the development of a number of guiding draft principles (e.g. from CBD, CIFOR, IUCN/CATIE, UNEP and UNFCCC) to assist practitioners in designing and implementing adaptation projects. A summary of shared principles that are emerging from different processes to characterize EbA include:

- Maintain ecosystem interactions and processes and promote the resilience of both ecosystems and societies, especially the most vulnerable.
- Support multi-sectoral approaches to adaptation as required by the cross-cutting nature and complexity of climate change and ecosystems.
- Consider multiple geographical and temporal scales and operate at scales that are appropriate to the objectives ensuring that management decisions are decentralized and taken at the lowest accountable level.
- Integrate flexible management structures and processes that enable adaptive management to anticipate and provide timely responses to changes.
- Recognize ecosystem limits and minimize negative social and environmental impacts and trade-offs.
- Use information and knowledge from multiple sources including traditional, local and scientific sources.
- Involve all stakeholders in the decision-making processes and balance their interests, in a participatory, transparent, accountable way.

Sources: adapted from CBD (2000), Heath *et al.* (2009), Colls *et al.* (2009), Andrade Pérez *et al.* (2010), Devisscher (2010), Glück *et al.* (2009), Doswald and Osti (2011), and UNEP (2012).

Climate variation and impacts: understanding the vulnerability context

Planning for EbA interventions usually starts with a vulnerability and risk assessment to identify who (i.e. specific population group) and what (i.e. ecological systems, human activities, infrastructure etc.) are particularly susceptible to the impact caused by climate variability and hazards (Magrin *et al.* 2007). These assessments include information on possible impacts on ecosystems and their implications for human well-being. Knowing the state of exposure and level of sensitivity of the socio-ecological system is important for planning EbA and can serve as a baseline for checking the effects of implemented adaptation inter-

ventions. Data can be collected using social field surveys in consultation with local stakeholders (e.g. through participatory rural appraisal tools) or environmental assessments (e.g. field inventories and satellite images) or a combination of both. According to a recent study, either method can be used for vulnerability assessments (Doswald *et al.* 2014).

Expected changes: choosing the right intervention

Information on current vulnerabilities can identify future climate situations and possible risks. Changes in the system's exposure to climate variations can alter its conditions, making some interventions unsuitable for species and habitats in the future. Taking into consideration the expected variations does not imply that we develop landscape interventions aimed at resisting or reverting changes by all possible means. Rather, EbA measures can help to facilitate ecosystem transitions and natural adaptation processes towards a new, socially acceptable conditions.

Ecosystem trade-offs: evaluating alternatives

Scenarios can include visions or projections of alternative situations depending on varying parameters and management choices that allow a comparison between EbA and other alternatives. Since EbA decisions may prioritize specific ecosystem services for adaptation interests at the expense of others, it is important to recognize and incorporate trade-offs that might occur. For example, a project targeting the stabilization of soil by revegetating slopes with exotic species might appear successful for that specific purpose, but it might provide relatively few products for local communities and increase the risk of fire. An adequate evaluation of ecosystem services would allow for a better comparison between the different ecosystem benefits and avoid undesirable side effects. Furthermore, the analysis of possible EbA effects should be extended to include technical alternatives with their costs and benefits. Such information are crucial for deciding which option should be undertaken and later can serve as a basis for designing interventions that balance human and ecosystem concerns, reducing the possibility of conflicts among different stakeholders. However, to date it seems that during the development of EbA, different options and social/environmental impact assessments are usually not considered (Doswald *et al.* 2014).

Progress towards ideal state: monitoring interventions and changes

Apart from methodologies and data availability, there are difficulties in measuring the impact of the interventions. This is partly due to the complex dynamics between people and ecological systems, which change over time and are greatly influenced by external inputs. In addition, ecosystems usually respond with weak signals during the duration of most projects and adaptation benefits are often recognized after several years. For these reasons, it is important that a monitoring and evaluation system is established to check progress and provide an indication

of changes so we can implement the necessary adjustments. A sound documentation and description of effective changes in the system can help to fill the gap in the scarcity of successful EbA experiences. Monitoring systems are currently found in the majority (65 per cent) of interventions related to EbA (Doswald *et al.* 2014).

Ecosystem-based adaptation and forest ecosystem service case studies

Comparing the experiences gained through the successful use of EbA in similar cases can help to select appropriate EbA measures. The next section includes case studies in which the adoption of EbA measures brought additional benefits related the topics addressed by the previous chapters. In particular, the three cases studies presented focus on implementing EbA measures to decrease the impacts on carbon stocks and timber (Chapter 6), water runoff (Chapters 3, 4 and 5), and recreation (Chapter 5) in tropical forests in Central America.

Ecosystem-based adaptation with benefits for carbon storage/mitigation

The reduction of carbon concentration in the atmosphere due to the ability of trees to absorb and store carbon in biomass is not the primary objective of EbA interventions. However, EbA increase the ability of the system to sequester and store carbon by improving overall ecosystem health and resilience. In addition, the possibilities offered by the carbon market can be exploited. For example the revenues gained from the sale of carbon credits can enhance the financial sustainability of ecosystem interventions for adaptation. Nevertheless, mitigation projects can have considerable transaction costs in order to estimate, validate and verify the carbon stored. To be profitable these projects need to be implemented on a large scale.

A case study on incentives for the conservation/restoration of ecosystem services: a Costa Rican payment for environmental services scheme

During the period 1950–1985, Costa Rica experienced a radical loss of forest cover, coupled with a demographic explosion (Rosero-Bixby and Palloni 1996). Forest cover declined by 50 per cent in this period, reaching 17–31 per cent mostly located on less accessible, steep mountain slopes (Sader and Joyce 1988; Lutz *et al.* 1993). During this 35-year cycle, the trend in land-use change has been driven, among other reasons, by the combination of market and policy incentives motivating landowners to expand crops and livestock (Lutz *et al.* 1993; Sanchez-Azofeifa *et al.* 2001).

National policies in the form of subsidies (forest payment titles), fiscal incentives, expropriation for protected areas and bans on land-use change in forest areas have stopped deforestation. Since 1996, Costa Rica adopted a new paradigm of compensating landowners to restore or maintain land uses that promise to provide ecosystem services of interest, such as: climate regulation, watershed

protection, landscape beauty and biodiversity conservation. The combined effects of payment for ecosystem services (PES) and the legal banning of land-use changes in forest lands allowed forest cover to reach 52.3 per cent of national territory in 2010 (FONAFIFO 2012).

Land-use activities such as forest conservation, reforestation, forest management, forest regeneration and agroforestry that are eligible for the Costa Rican PES scheme contribute to both climate change mitigation and adaptation objectives. These practices have positive effects on the conservation or enhancement of tree biomass[1] and protection of water resources, while helping to conserve biodiversity that can have a value for adaptation (Box 8.2).

Box 8.2 Possible ecosystem-based adaptation interventions with mitigation benefits

Afforestation and revegetation of degraded lands:

- Enhance forest recovery after natural or man-made disturbances.
- Decrease rotation periods for coppices used for energy production.
- Establish new forests areas through planting or assisted regeneration.
- Facilitate natural expansion of forests.
- Increase tree cover density in agricultural lands (e.g. agroforestry).

Reducing forest degradation and avoiding deforestation:

- Practice low-intensity forestry and low-impact logging and prevent conversion of primary forests to plantations.
- Work with authorities and stakeholders to address the causes and drivers of deforestation.
- Ensure law enforcement for illegal logging and NTFPs.
- Implement a sustainable level of firewood collection.

Sustainable management:

- Modify thinning practices (timing, intensity) and rotation length to increase growth and turnover of carbon.
- Promote research and cooperation to increase the socio-ecological system understanding.
- Minimize soil disturbance through low-impact harvesting activities.
- Manage forests fires and pests to decrease the impact of natural disturbances on carbon stocks.

Conservation:

- Preserve ecological process while maintaining access to ecosystems.
- Protect through protected area systems.

Sources: adapted from CBD (2009), Innes *et al.* (2009), FAO (2013), and UNEP (2012).

The linkages between mitigation and adaptation have become clearer in the 18-year-long experience of the Costa Rican PES scheme. Indeed, this scheme started with an explicit recognition of carbon benefits provided by the considered land-use activities to evolve in the formal mechanism for supporting the REDD+[2] goals. With the REDD+ Readiness Proposal of 2010 we find a formal recognition of the EbA benefits provided by those land-use activities to society (GoCR 2011). The prioritization criteria for the selection of beneficiaries listed by FONAFIFO have evolved based on experiences gained through the implementation of the scheme. These criteria have evolved from a general conservation-oriented approach defined at local level to a more refined approach, responding to national priorities and to the applicants' location (in term of socioeconomic and environmental aspects) and land-use context (Porras *et al.* 2013). These criteria guide the selection of areas where incentives are targeted and help the scheme by providing important benefits to both mitigation and adaptation (Table 8.1).

The Costa Rican PES scheme has stimulated the recent literature to identify how to improve the design of the PES scheme in order to increase its synergism with EbA objectives and offer institutional support. Some literature has looked at additionality issues (Pfaff *et al.* 2008), highlighting that many areas targeted for conservation payments had low opportunity cost of land (i.e. the amount of the payment is not attractive for areas with higher opportunity cost), which may not be the most suitable for achieving EbA goals. The challenge remains for ecosystem services-conservation efforts to ensure that EbA-relevant areas are targeted, regardless of their opportunity costs. Other authors have referred to the effectiveness and conditionality aspects of the PES scheme, indicating that targeted land uses (and management alternatives) might not deliver the required ecosystem services (Wunder *et al.* 2008). For example, certain forest management practices aimed to protect watersheds and increase biological connectivity can have unforeseen side effects. This can make it difficult to predict their effectiveness, as site conditions determining their function might change with changing climatic and hydrological conditions (Hagerman and Chan 2009; IIED 2011).

Ecosystem-based adaptation with benefits for water runoff regulation

One of the consequences of climate variability is the increased frequency in the disruption of water availability, causing either a shortage or an excess. Trees and vegetation help to regulate water runoff and river discharge during periodic interruptions in seasonal rainfall, help to improve the water quality and retention, and to buffer against coastal damage from tropical storms and tsunamis. There are a limited number of studies that support the important role of forested landscapes in regulating these processes (Pattanayak and Kramer 2001; Ilstedt *et al.* 2007) and generalizations are difficult to make (Bruijnzeel 2004). In fact, the total contribution of trees in regulating the water cycle is a result of complex processes and interactions that depend on a number of factors such as: land-use practices, topography, vegetation types, soil properties, and the intensity of the hazard. In particular, the scientific debate remains on the protective potential of vegetation

Table 8.1 Linkages between forest conservation activities promoted by Costa Rican PES scheme and forest adaptation and mitigation

Activity funded	*FONAFIFO prioritization criteria*	*Rationale*	*Potential benefits for adaptation*	*Potential benefits for mitigation*
Forest conservation	Forest located in conservation gaps	Biodiversity conservation	Adaptation of forest ecosystems through addressing conservation gaps in biological corridors to increase landscape connectivity and reduce fragmentation (Locatelli *et al.* 2011)	Conservation/restoration of tree biomass
	Forest located within biological corridors			
	Forest located in areas relevant for protection of water resources	Water quality and regulation	EbA benefits forest biomass conservation/ restoration and the future supply and/or quality of water ecosystem services	Conservation/restoration of tree biomass
	Forest located in indigenous territories	Multiple ecosystem services considered	Forest conservation can provide important adaptation benefits to indigenous communities helping to protect their livelihoods and culture	Conservation/restoration of tree biomass
Reforestation	Reforestation in highly productive sites	Ecosystem restoration and sustainable use	Adaptation benefits can be achieved since reducing pressures on ecosystems increases ecosystem resilience in the face of climate change (Malhi *et al.* 2008)	Enhances carbon stocks by increasing tree biomass
	Reforestation with native species or species facing extinction			
Forest management	Forest management in highly productive sites using sustainability standards			
Agroforestry	Promoting agroforestry in highly productive sites			

in the case of extreme floods and storms (Osti *et al.* 2009). In addition to water regulating services, vegetation coverage protects against erosion, reduces soil loss and transportation of sediments and debris (which if mixed with floodwater, increases its destructive power and reduces water quality). Therefore the role of forested landscapes in preventing average and most frequent floods should not be overlooked.

The case of water runoff related ecosystem services in Honduras

Honduras is highly vulnerable to the impact of climate change especially due to:

- the observed and projected impacts of extreme rainfall events (Aguilar *et al.* 2005; Magrin *et al.* 2007);
- the sensitivity of its agricultural production systems that are located in high sloping areas (Perotto-Baldiviezo *et al.* 2004); and
- the high per centage of its population living in extreme poverty (especially in these sensitive areas).

In such a context, soil laminar erosion is a serious threat to upstream marginal farmers affected by soil fertility degradation (Holt-Gimennez *et al.* 2001) and to downstream users of water resources that can be polluted by transported sediment. Here, ecosystem-based responses such as watershed conservation efforts to reduce soil erosion and water infiltration can be important in reducing the impacts of extreme runoff, and help the conservation of water-related services such as those associated with water quality under climate change.

Guacerique Watershed, which provides an important share of the drinking water to the capital city of Tegucigalpa, illustrates how EbA options can reduce the costs of water delivery and increase water quality in the face of climate change. Guacerique Watershed is a mountainous region with a mean altitude of 1450m and the majority of the land area is comprised of slopes greater than 15 degrees (SANAA and ICF 2011). These sloping areas are best suited for forest land uses but many are under subsistence agriculture farms, of which only 43 per cent have carried out soil and water conservation measures (TroFCCA 2009). The upper section of the watershed has been declared a biological reserve (Decree 87-1987) managed by the Secretary of Natural Resources (SERNA) and municipalities. Guacerique Watershed has undergone significant changes in land use in recent decades, including deforestation and conversion to agriculture (even within the boundaries of the biological reserve) and human settlement (Shlomo *et al.* 2004), resulting in an annual aggregate rate of deforestation for the period 1993–2008 of 1.36 per cent (TroFCCA 2008).[3]

The pressure on ecosystems' ability to provide goods and services combine with the pressure of climate change, as indicated by projected increases in temperature and uncertain annual precipitation trends ranging from a reduction of 34 per cent to an increase of 9 per cent by 2080 (Vignola *et al.* 2013). This vulnerability which threatens water resources supply is aggravated by the projections of population

Box 8.3 Possible ecosystem-based adaptation interventions with water regulation benefits

Regulation of water flows and provision of clean water:

- Maintain forests on crests and steep slopes to promote mist and fog interception.
- Protect forests in water catchment areas and consider excluding harvesting in areas subject to waterlogging.
- Select water-efficient and drought-resistant species and varieties for afforestation and reforestation.
- Carry out vegetation management (e.g. weed control) to limit adverse hydrological effects.
- Reduce evapotranspiration and competition for water by vegetation management (e.g. thinning, pruning and planting deciduous species).
- Promote multilayered root systems by encouraging growth (e.g. through natural regeneration or planting) of deep-rooted and shallow-rooted species.

Buffer effect of water fluctuations:

- Ensure unimpaired water flow by removing debris and blockages (stones, logs, waste) that could increase hazards to people.
- Consider reverse channelization through renaturation interventions, in particular: increase natural vegetation to absorb impacts of floods and block sudden storm surges and incursions of seawater (for coastal and marine ecosystems) and around river systems to provide space for floodwaters.
- Leave a buffer strip of forest between a stream or river and any area used for forestry operations.
- Protect peatlands.
- Encourage species and varieties capable of benefiting from or withstanding increased rainfall and waterlogging.

Stabilization of soil:

- Decrease possibilities of sediment runoff (stabilization).
- Minimize soil disturbance and avoid soil compaction with low impact harvesting techniques and timing.
- Plan road construction carefully.
- Promote afforestation and reforestation to protect against wind erosion (e.g. establish windbreaks).
- Stabilize sand dunes and desert margins in areas affected by desertification.
- Adjust harvesting schedules to reduce erosion and siltation.
- Maintain or increase vegetation cover.

Sources: adapted from CBD (2009), Innes *et al.* (2009), FAO (2013), and UNEP (2012).

increase in Tegucigalpa (expected to double its population by 2030) and its increased demand for drinking water, which will be 4.52 m^3/s of water stream flow, compared to 1.8 m^3/s available today (SOGREAH 2004).

In the face of these complex challenges, local authorities are implementing an Adaptation Fund project which aims to design and implement a robust watershed management plan to reduce current and future vulnerabilities under the supervision of the National Water Utility (SANAA) and the Honduran Ministry of Forests (ICF). This plan aims to ensure long-term water availability and to lower sediment loads in the Guacerique River in order to maximize the watershed's utility as a source of drinking water for Tegucigalpa (SANAA and ICF 2011).[4] The six-year watershed management plan costs US$4,216,000 (in 2012) and includes both environmental and poverty alleviation objectives. It will implement activities such as:

- reforesting 1236 ha around springs and creeks;
- creating 100 ha of fuelwood plantations;
- transitioning to agroforestry on 161 ha of steeply sloping agricultural land (on slopes of 30 degrees or more);
- concentrating forest fire control on reforested areas;
- reducing illegal timber extraction on 6063 ha classified as forest reserve;
- concentrating pest control on 4338 ha of existing pine forests; and
- implementing soil and water conservation measures on 2000 ha of agricultural fields.

SANAA is already implementing an agricultural extension project in the watershed to encourage adoption of soil retention measures by local producers, in the hope that a compensation mechanism will help to sustain these soil conservation activities. Similarly, several initiatives are already being implemented to promote the conservation of the Guacerique Watershed including conservation planning with local communities on their micro-watersheds, solving legal problems related to tenure in biological corridors, and installing micro-irrigation systems to increase water-use efficiency in farming. A more extensive set of possible EbA interventions with water regulation benefits is presented in Box 8.3. Even if the financial sustainability of this plan (after the adaptation fund start-up costs) is an ongoing challenge, some monetary benefits can already be estimated. A recent analysis shows that the watershed management plan can have important benefits as measured by: sedimentation rates, turbidity, dissolved oxygen and water inflow (Vignola *et al.* 2013). The monetization of these benefits suggests that the overall annual economic benefit of the watershed management plan to the national water utility for 2030 to 2035 (expressed in undiscounted 2012 dollars) ranges from US$3.7 million (under a less pessimistic scenario) to US$9.2 million (under a pessimistic scenario). The net economic benefit expected to accrue from the watershed management plan ranges (depending on the social discount rate used) from US$23.6 million to US$34.7 million for the less pessimistic climate change scenario and from US$63.6 million to US$91.5 million for the pessimistic

one. These economic benefits only estimate the value of the watershed management plan for drinking water, while several other benefits, as indicated by local experts, can be provided by conserving Guacerique Watershed. These should be included in an overall assessment of the benefits of EbA options, if a cost-benefit analysis is required to compare with the benefits of infrastructure. Indeed, at least in the Guacerique case, infrastructure alternatives to deal with the challenges of drinking water provision to Tegucigalpa have been discussed and feasibility studies have been developed but to date, it has been impossible to implement them, given their high costs (over a hundred million US dollars; Vignola *et al.* 2013). The EbA option in this case is a relatively cheap and available alternative. This confirms its advantages in helping society to design and implement short-term responses to severe and complex problems related to ecosystems' goods and services degradation due to climate change.

In order to use this EbA alternative to achieve sustainable provision of drinking water to Tegucigalpa in the face of climate change, some challenges and barriers must be addressed (Vignola *et al.* 2013). National experts identified these as a need to ensure an adequate flow of financial resources and a more efficient use of scarce resources through inter-organizational coordination. They also outlined the importance of a combination of incentives mechanisms for good practices and more patrolling for control of illegal uses of the watershed's ecosystems. They highlighted the need for more effective communication mechanisms to improve information-sharing from monitoring systems. They outlined the existence of enabling policies for PES schemes; the importance of forest ecosystems' conservation and sustainable use in national forest and climate change policies; and the important role assigned to local communities and municipalities in managing their ecosystems.

Ecosystem-based adaptation with benefits for recreation values

EbA interventions do not directly aim to increase recreation opportunities. However, by using green solutions instead of concrete/steel infrastructure for adaptation, positive co-benefits in term of visual impact and attractiveness might occur. Due to cultural differences and preferences in how forest aesthetics are considered, it is difficult to determine how adaptation measures help to maintain or increase the aesthetic value associated with landscapes. Nevertheless, a loss of habitat and diversity and the consequent simplification of landscapes often has a major impact on the attractiveness of a place in terms of recreational use.

A case study on ecosystem-based adaptation options for ecosystems' recreation services in Central America

The relationship between nature-related tourism and the impacts of climate change on associated ecosystems is increasing our awareness of the importance of ecosystems' goods and services preservation to maintain recreation services, which represent, especially in Central America, an important economic sector. However,

efforts focusing on EbA options to conserve ecosystems' recreation services in the face of climate change are still in their infancy in the region. On the other hand, there are interesting experiences in the region in designing and implementing sustainable tourism initiatives related to ecosystems' recreation services which can represent ecosystem-based cost-effective measures providing multiple co-benefits among which the potential to contribute to climate change adaptation (Box 8.4). In Panama, ecotourism has grown by more than 100 per cent over the last decade (Christ *et al.* 2003). Community-based ecotourism activities represent an important share (10 per cent) of national GDP and have been analyzed over the past 15 years (ATP 2011). The barriers and priorities identified by the experiences in Panama can be relevant for designing EbA options for ecosystems' recreation services. Among the most important tourism assets in Panama (in terms of quality, variety and potential) are protected areas (representing 73 per cent of the national tourism attraction asset; Análisi Diagnostico General Del Turismo En Panama 2008) where indigenous groups make up the majority of the residing population. Many of these natural habitats are already threatened by extreme climatic events such as hurricanes and tropical storms, which can cause forest loss and damage to tourist infrastructure and roads (CEPAL 2010).

Box 8.4 Possible ecosystem-based adaptation interventions with recreation benefits

- Include traditional knowledge about climate change into planning.
- Expand tourism and recreational services to multiple season operations.
- Include information about natural and cultural heritage values in the decision-making processes.
- Promote natural and cultural heritage values through the marketing of products based on local ecosystem goods and traditional practices.

Source: adapted from CBD (2009), Innes *et al.* (2009), FAO (2013), and UNEP (2012).

Community-based ecotourism can protect ecosystems' recreation services in areas that are vulnerable to climate change, while at the same time providing opportunities to empower local communities and diversify their livelihood strategies, which can lead to an increased capacity to cope with, and recover from, climate impacts.

Consultations held with communities of the Panama Canal (Lumpkin 1998) highlighted the importance of their involvement in ecotourism and provided interesting examples for EbA promotion in similar contexts. These communities are, as in the case of Embera indigenous communities, already implementing, actions to value their culture in relation to ethnic/cultural benefits for eco-tourists. Communities mentioned traditional uses of many non-timber forest products (e.g.

medicinal plants and domesticated wildlife in home yards) and timber products for traditional construction (e.g. canoes, thatched-roofed huts) as assets for community ecotourism development. In these consultations, equity gains were also highlighted, such as the empowerment of women through their role as providing community hospitality for tourists and the opportunity to create corresponding women-run small enterprises. The inclusion of former illegal hunters as ecosystem conservation guardians or as ecotourism guides enhanced the effectiveness (and acceptance) of these ecosystem-based options for community development, while smoothing out potential conflicts and unequal distribution of tourism benefits[5] and making use of traditional knowledge developed by these hunters.

On the other hand, important barriers hamper the capacity of potentially interested communities to participate in, and benefit from, recreation services. One important barrier to the effective achievements of benefits from these initiatives is the poor state of infrastructure that allows access to these habitats during the rainy season and especially during extreme rainfall events. There are no reported efforts to plan adaptation measures for poor road infrastructure, even if the intensity and frequency of extreme rainfall is expected to increase. An additional limitation mentioned by local communities is the national legislation that prohibits the local use of some resources (e.g. timber and some protected species) posing high transaction costs to harvest material for traditional construction.

Many of the benefits of ecosystems' recreation services identified during local consultations are relevant for community development purposes and for specific adaptation concerns. Indeed, community empowerment, addressing equity concerns and valuing traditional knowledge and lifestyles as in the case of Panama, can increase the capacity of communities to negotiate adaptation plans and enhance their sense of place and territorial identity. Furthermore, such actions can consolidate communities' abilities and motivation to care about habitat conservation, which is necessary for planning, implementing and sustaining ecosystem-based recreation services strategies with community ownership (Plummer and Fennel 2009; Strickland-Munro *et al.* 2010).

In 2012 Panama was rated first in the *New York Times*'s list of tourist destinations (Williams 2012). If ecotourism is to maintain its sustainability promises, communities must be directly engaged in the design and management of natural habitats and related tourist facilities. In order to do so, enabling activities should be promoted by national policies such as improving: communities' education and capacity to run tourist businesses; road infrastructure; and communication technologies (e.g. facilitating access to the Internet) (Eddins 2013). Important initiatives (e.g. from the Inter-American Development Bank) are to promote low-impact ecotourism within Panama's national system of protected areas (SINAP) on the basis of innovation, participation of local businesses and local sustainable social development.

The case of Panama's ecosystems recreation services for ecotourism highlights a couple of key relevant lessons for tropical forest ecosystems and indigenous culture and livelihoods. In order to ensure that the EbA benefits are accrued by

vulnerable populations, attention should be paid to conservation efforts; infrastructure building and maintenance; business education and empowerment; revision of legal settings allowing for sustainable use of local ecosystems' goods and services; among other things.

Conclusion

Understanding climate impacts and possible adaptations in socio-ecological systems is essential to avoid, or plan for, undesired changes and their negative repercussions on human well-being. Knowledge of socio-ecological systems and their dynamics provide a fundamental basis for developing appropriate adaptation interventions that are locally based and relevant for extended time scales. In order to reduce people's vulnerability, interventions should focus on enhancing adaptive capacity (e.g. availability of alternative livelihoods, removal of practices that increase pressure on ecosystems, adoption of appropriate technologies and good governance), decreasing exposure (e.g. buffers between system and climate hazards), or decreasing sensitivity (e.g. dependency on a few, weather-prone productive activities). Adaptation strategies based on the use of ecosystems can be used to address the complex linkages between climate change and people's well-being, through ecosystems' ability to provide goods and services relevant for adaptation and beyond. EbA often offers a set of options that is immediate, flexible and cost-effective, and complements other technical and development interventions.

Many Central American countries acknowledge the role of ecosystem services in their policies because of the relevance that natural resources have in their economies. Several approaches have been designed and are implemented in the region, including PES, protected areas, and biological corridors. These landscape interventions can strengthen the delivery of important ecosystem services for biomass, water and recreation services. However, the region is only beginning to acknowledge the adaptation values of these policies to promote ecosystem-based alternatives to reduce the impacts of climate change.

Many of the barriers that hamper regional advances in EbA are related to (i) those identified in the literature evaluating the performance of PES schemes and (ii) those related to the evolution of the climate change agenda. An example of the first set of barriers is represented by the scarcity of data and knowledge of the relationship between different land uses and management options and the provision of ecosystem services (i.e. the conditionality requirement of PES schemes), which might reduce the capacity of institutions to design effective and efficient ecosystem service conservation schemes. The second set of barriers is represented by how, only recently, the regional agenda has been moving towards adaptation efforts by examining synergies with land-use mitigation efforts (e.g. in REDD+). Following the Rio Conference in 1992, the land-use-based responses to climate change in the region focused on the climate regulation role of forest ecosystems, with acknowledgement of the potential positive development externalities of these mitigation initiatives. The recent recognition of the failure of land use, land-use change and forestry (LULUCF) initiatives in the region has

led to the acknowledgement of a wide range of ecosystem services which play an important role in maintaining productive activities, leisure and spiritual goods for society in the face of climate change.

The first Central American Conference of Ecosystem-Based Adaptation was attended by more than seventy participants (from NGOs, community-based organization, government agencies, multilateral and scientific organizations) and outlined ways of fostering nature-based adaptation alternatives (Vignola *et al.* 2009). Most of the messages targeted the role of NGOs, scientists and policy-makers in designing and implementing EbA alternatives, highlighting the importance of strengthening institutional coordination and communication mechanisms, and effective community participation. This reinforces the issues highlighted in the cases presented here. Integrating EbA in promising initiatives in the region can help to reconcile different aims in sectoral policies and achieve multiple objectives in a cost-effective way for more resilient societies and environments.

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Notes

1 During 2000–2005, forest regeneration mitigated by sequestration around 11.2 Gg of CO_2 per year (GoCR 2011), which represents more than double 2005's CO_2 emissions of energy sector (Chacón *et al.* 2009).

2 Reducing Emissions from Deforestation and Forest Degradation.

3 However, TroFCCA (2008) reports disaggregated rates for mixed and conifer forests for the same period as high as 2.8%, which are higher than the national mean recorded in other studies such as Rivera (1998).

4 As mandated by the Law on Forests, Protected Areas and Wildlife, forest-related management plans, including watershed management plans, are the responsibility of ICF. However, the watershed management plan for the Guacerique Watershed has been a collaborative effort between SANAA and ICF and responsibility for implementation has been delegated to SANAA.

5 This unequal distribution is associated with the exclusion of this part of the population whose livelihoods based on wildlife would have been affected.

Index

"This is the definitive volume of readings on the climate change and ecosystems services in the forests of Central America. The readings provide viable insight and analysis, and should serve as inspiration for researchers not only interested in this problem but also in conducting studies of climate change impacts on ecosystems in other critical regions of the world."–Edward B. Barbier, University of Wyoming, USA

"Tropical forests are among the most valuable ecosystems on earth in terms of their biodiversity and their contribution to sustainable human well-being. This book is a timely and valuable addition to the growing literature on tropical forest ecosystem services."–Robert Costanza, The Australian National University, Australia

"This is a worth reading volume for researchers and policy makers which presents one of the few ecological and economic comprehensive studies of how climate change is affecting ecosystem services. The methodological approach and analysis, combining different types of spatially explicit data within various models and scenarios, should serve as a model for other critical regions."–Maria Jose Sanz Sanchez, Agriculture Organization of the United Nations (FAO), Italy

This book addresses the impacts of climate and land-use change on tropical forest ecosystems in Central America, and assesses the expected economic costs if no policy action is taken, under different future scenarios and for different geographical scales.

This innovative collection utilises both theoretical approaches and empirical results to provide a conceptual framework for an integrated analysis of climate and land-use change impacts on forest ecosystems and related economic effects, offering insight into the complex relationship between ecosystems and benefits to humans. This important contribution to forest ecosystems and climate change provides invaluable reading for students and scholars in the fields of environmental and ecological economics, environmental science and forestry, natural resource management, agriculture and climate change.

Aline Chiabai is a senior researcher at the Basque Centre for Climate Change (BC3), where she coordinates the research area on health and climate change.

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